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United States Department of Agriculture
Natural Resources Conservation Service

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Agricultural Research Service

SRM Society for Range Management

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DAVID D. BRISKE

Conservation Benefits of Rangeland Practices

Assessment, Recommendations, and Knowledge Gaps

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DAVID D. BRISKE, EDITOR

On the cover: "Zion Sunset" highlights several famous land forms, including West Temple, Altar of Sacrifice and the Sentinel in Zion National Park in southern Utah. Photographer: Steve Guymon, Providence, Utah; www.steveguymonphotograpy.com.

United States Department of Agriculture
Natural Resources Conservation Service

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Society for Range Management

The expressed goal of this synthesis is to inform deliberations of managers and policymakers regarding the current effectiveness and potential improvements to rangeland conservation practices and programs. As such, this synthesis represents a scientific assessment that was reached independently of the current position or policy of the US Department of Agriculture or the United States government.

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ABOUT NRCS: HELPING PEOPLE HELP THE LAND

The Natural Resources Conservation Service (NRCS) is a USDA agency that works with landowners through conservation planning and assistance to benefit the soil, water, air, plants, and animals for productive lands and healthy ecosystems. Working in nearly every county in the Nation, NRCS employees understand local resource concerns and challenges to assist with conservation solutions that last. Land stewardship by private landowners is critical to the health of our Nation's environment. Science and technology are critical to good conservation. NRCS experts from many disciplines come together to help landowners conserve natural resources in efficient, smart and sustainable ways. NRCS succeeds through partnerships, working closely with individual farmers and ranchers, landowners, local conservation districts, government agencies, Tribes, Earth Team volunteers and many other people and groups that care about the quality of America's natural resources. More information about the NRCS is at www.nrcs.usda.gov.

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ABOUT SRM: SOCIETY FOR RANGE MANAGEMENT

The Society for Range Management is the professional society dedicated to supporting persons who work with rangelands and have a commitment to their sustainable use. Rangelands comprise almost one-half of all the lands in the world. They are extremely important to society for the goods and services they produce and for the ecological services they provide. SRM is dedicated to the conservation and sustainable management of rangelands for the benefit of current societies and for future generations. More information about the SRM is at www.rangelands.org.

Society for Range Management

Conservation Benefits of Rangeland Practices

Assessment, Recommendations, and Knowledge Gaps

> This document was written to provide an unprecedented source of evidence-based information to guide the development and assessment of management practices and conservation programs on the nation's rangelands.

Leonard W. Jolley
David D. Briske

Conservation Benefits of Rangeland Practices: Assessment, Recommendations, and Knowledge Gaps

United States Department of Agriculture, Natural Resources Conservation Service

David D. Briske, editor

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PREFACE

Conservation is a major tenet of American society that is symbolized by Aldo Leopold's iconic work *Sand County Almanac*. A conservation ethic emphasizes the protection, management, and restoration of natural resources for the public benefit, including sustainable social and economic utilization. The importance of conservation was imprinted on the national psyche by several episodes of environmental degradation, including severe overgrazing of western rangelands in the late 19th century and the Dust Bowl of the 1930s—the latter of which provided the impetus for organization of the current-day Natural Resource Conservation Service. Black Sunday—a severe dust storm that occurred in the southern Great Plains on April 14, 1935—serves as a symbol of the devastating consequences of unsustainable land use on both natural resources and human well-being that is dependent on them. The rangeland profession similarly emerged from the actions of early government researchers and managers, focused largely on grazing management and restoration, to halt and reverse degradation of western rangelands in the late 19th century. These episodes of natural resource degradation have contributed to the growing awareness that conservation of the nation's natural resources is as much about managing human actions and values as it is about managing natural resources themselves.

In spite of broad recognition of the importance of natural resource conservation to the nation, it is necessary to substantiate the outcomes of conservation programs in an era of increasing fiscal responsibility and accountability. The Conservation Effects Assessment Project (CEAP) was created to assemble the baseline knowledge of rangeland conservation programs, inspire innovation in the design and implementation of future programs, and provide a blueprint for the delivery of science-based and cost-effective conservation programs. CEAP expressly emphasizes that conservation programs address the environmental quality of lands in addition to the sustainable production of agricultural goods. Future conservation programs will be increasingly called on to evaluate the benefits of local agricultural production relative to the maintenance of regional ecosystem services.

The academic community has embraced the vision of CEAP and has committed to this synthesis by retrieving and evaluating thousands of published research papers and compiling the most relevant information into readily accessible forms. The evidence-based recommendations originating from this synthesis can guide the development and assessment of future management practices and conservation programs. The knowledge gaps that have been identified can inform funding programs of areas in need of further research. Success of the Rangeland CEAP Synthesis will partially be determined by 1) the extent to which it can strengthen the linkage between scientific and management knowledge, 2) advance conservation science and policy, and 3) promote assessment of societal benefits, including both agricultural goods and ecosystems services, emerging from conservation programs.

Even though the Rangeland CEAP Synthesis was explicitly designed and implemented to assess conservation programs of the US Department of Agriculture–Natural Resources Conservation Service, it has broad and significant implications to the entire rangeland profession. This synthesis provides a compelling argument for the development of an expanded rangeland research agenda that can more effectively articulate and embrace the scope and complexity of the conservation challenges that are most pressing to the nation.

Rangelands are complex adaptive systems that encompass both ecological and social components as well as the intricate and poorly understood interactions among these components. This requires that social knowledge of rangeland systems, including management, socioeconomics, and policy, merit equal priority to that of ecological knowledge because they collectively establish conservation success. Therefore, conservation programs and practices within rangeland systems should be designed, implemented, and modified on the basis of multiple knowledge sources acquired from both anticipated and unanticipated conservation outcomes. Partnerships among natural resource managers, researchers, and policymakers are likely to generate the most relevant knowledge to address the emerging conservation challenges confronting the nation.

David D. Briske
Editor and Academic Coordinator
Rangeland CEAP Synthesis

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I wish to acknowledge the commitment, skill, and persistent diplomacy on the part of our academic coordinator and document editor Dr. David D. Briske, Texas A&M University. The success of this publication would be very much in doubt without his capable leadership and consistent vision for the potential of Rangeland CEAP. Dr. Tom Thurow, University of Wyoming, conducted a thorough review of the entire draft document to assist with matters of tone, content, consistency, and context. His insight and passion for the project are greatly appreciated.

Special recognition must be afforded the authors since much was asked of these subject matter experts, and most of them labor in an environment where synthesis documents are not very helpful on matters of promotion and advancement in the research world. They are a very talented group, and their labors on this milestone document are greatly appreciated. Nearly 30 external reviewers were involved in the evaluation of this document, and their significant inputs enhanced the quality of this work.

The US Department of Agriculture–Natural Resources Conservation Service (USDA-NRCS) and its leadership are to be commended for initiating and consistently supporting this project to scrutinize the science supporting rangeland conservation practices. The CEAP Steering Committee was a consistent advocate for the pursuit and publication of this literature synthesis. I extent a special thanks to the NRCS specialists who functioned interactively with the science writing teams to ensure that the use of conservation practices in the NRCS planning environment was fully appreciated.

The Agricultural Research Service (ARS) allowed several of their scientists to function as authors on this project. The long-term nature of the ARS research vision and their selfless research and service contributions are truly impressive. Dr. Mark Weltz (USDA-ARS) was crucial to the administration of key agreements in support of this project. The Society for Range Management supported Rangeland CEAP by hosting an initial organizational meeting at their national headquarters and by managing USDA funds to conduct this project.

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(Photo: Chris Call, Invasive species management; p. 321)

> The recent success of CEAP provides numerous opportunities and challenges to achieve its full potential within the USDA and the broader conservation community

Introduction to the Conservation Effects Assessment Project and the Rangeland Literature Synthesis

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THE CONSERVATION EFFECTS ASSESSMENT PROJECT

The Conservation Effects Assessment Project (CEAP) is a unique, multiagency effort designed to quantify conservation effects and to determine how conservation practices can be most effectively designed and implemented to protect and enhance environmental quality (Duriancik et al. 2008). CEAP was jointly initiated in 2003 by the Natural Resources Conservation Service (NRCS) in partnership with the Agricultural Research Service (ARS) and the National Institute of Food and Agriculture (NIFA) in response to requests from Congress and the Office of Management and Budget for greater accountability to US taxpayers following a near doubling of US Department of Agriculture (USDA) conservation program funding in the 2002 Farm Bill. These funds are allocated to multiple conservation practices through several USDA-sponsored conservation programs, including the Environmental Quality Incentives Program, Wetlands Reserve Program, Wildlife Habitat Incentives Program, Conservation Reserve Program, and NRCS Conservation Technical Assistance Program. This funding increase was concomitant with substantial modifications to conservation programming that emphasized *environmental quality* of these lands in addition to sustainable agricultural production (Mausbach and Dedrick 2004).

CEAP Goals

The primary goal of CEAP is to strengthen the scientific foundation underpinning conservation programs to protect and enhance environmental

Rangelands represent non-cultivated, non-forested land that is extensively managed with ecological principles. (Photo: David Briske)

Rangelands are comprised of diverse and extensive ecosystems that provide multiple goods and services to society. (Photo: Rick Miller)

quality of managed lands. CEAP is focused on establishing principles to guide cost-effective conservation practices at landscape scales and to achieve multiple environmental quality goals by placing specified conservation practices or combinations of complementary practices at appropriate locations on the landscape to maximize their effectiveness. CEAP is also developing science-based guidance, information, and decision support tools to determine the appropriate practices to be implemented at various locations on the landscape and to provide conservation program managers with a blueprint for delivery of science-based and cost-effective conservation programs (Duriancik et al. 2008).

A secondary goal of CEAP is to establish a framework for assessing and reporting the full suite of ecosystem services impacted by various conservation practices. Ecosystem services represent the benefits that ecological processes convey to human societies and the natural environment. For example, agricultural lands provide flood and drought mitigation, water and air purification, biodiversity, carbon sequestration, nutrient cycling, and aesthetics and recreation, in addition to the primary agricultural commodities produced. These ecosystem services are often taken for granted and unpriced or underpriced by the marketplace. Research and assessment activities will be integrated within CEAP to provide a scientific foundation for assessing the extent to which ecosystem services are enhanced by conservation practices and programs.

Organization and Approach

The USDA engaged the Soil and Water Conservation Society in 2005 to assemble a panel of university scientists and conservation community leaders to recommend the most effective, proactive, and scientifically credible CEAP activities—thereby ensuring that

CEAP products would have wide utility for diverse stakeholders within the conservation community. CEAP has evolved into an assessment and research initiative directed at determining not only the impacts of conservation practices, but also evaluating procedures to more effectively manage agricultural landscapes in order to address environmental quality goals at local, regional, and national scales (Maresch et al. 2008).

Three principal themes will guide CEAP investments and activities in the future (Maresch et al. 2008):

1. Research addressing effective and efficient implementation of conservation practices and programs to meet environmental goals and enhance environmental quality.

    - Continue and expand CEAP research projects on the effects and benefits of conservation practices for soil and water quality at the watershed and landscape scales.
    - Implement a new research and assessment initiative for grazing lands designed to provide scientific evidence for implementation of conservation practices at the landscape scale.
    - Determine the critical processes and attributes to be measured at the appropriate landscape position for evaluation of environmental benefits.
    - Expand the scope of assessment to include evaluation of a full suite of ecosystem services influenced by conservation practices and programs.

2. Assessment of the environmental impacts of conservation practices for reporting at the regional and national scales.

    - Continue CEAP activities designed to estimate environmental benefits of conservation practices and programs.
    - Develop a framework for reporting impacts of conservation practices and programs in terms of ecosystem services.
    - Identify future conservation requirements and provide information for setting national and regional priorities.
    - Expand assessment capabilities to address potential impacts of changes in agricultural land use and policy and define necessary conservation programs to meet new environmental challenges brought about by alternative land use or policy changes.

3. Translation of science into practice by developing a blueprint for integrating scientific knowledge into the conservation planning and protocols for implementation.

    - Communicate research findings and lessons learned about managing agricultural landscapes to a broad audience.
    - Develop strategies for communicating scientific findings and recommendations to farmers, ranchers, and NRCS field office staff describing opportunities to enhance environmental quality.
    - Conduct studies to determine the types of tools and resources field offices require to evaluate and implement conservation practices within landscapes.
    - Conduct studies to demonstrate effective implementation of landscape management and adaptive management to conservation planning, implementation, and monitoring.
    - Develop tools that can be used by NRCS field offices to identify the most appropriate practices to be applied at the most appropriate landscape positions to effectively and efficiently meet local and regional environmental goals.

CEAP has been organized into four national assessments addressing croplands, wetlands, wildlife, and grazing lands—grazing lands are subdivided into rangelands and pasturelands based on distinct climate and management considerations. CEAP also includes several watershed investigations, which are sponsored by ARS, NIFA, and NRCS. These studies provide quantitative science-based outcomes and methodological innovations to support the national assessments. Bibliographies and literature syntheses have been developed for

“CEAP has been organized into four National Assessments addressing croplands, wetlands, wildlife, and grazing lands—grazing lands are subdivided into rangelands and pasturelands based on distinct climate and management considerations”

TABLE 1. Areas treated by each of the seven major conservation practices addressed in the rangeland synthesis in the west and central regions of the United States during 2004–2008.

Conservation practice	USDA code	Area (ha)
Prescribed grazing	528	31 359 980
Prescribed burning	338	370 821
Brush management	314	1 456 837
Range planting	550	517 301
Riparian herbaceous cover	390	12 352
Upland wildlife habitat management	645	19 165 668
Pest management	595	7 603 070

each of the four national assessments and the watershed assessment; these document the current state of knowledge regarding the effectiveness of conservations practices, provide recommendations to enhance conservation programs, and identify critical knowledge gaps that require further research (Duriancik et al. 2008).

RANGELAND CEAP

Rangeland CEAP was formally initiated in 2006 to evaluate conservation effectiveness on rangelands (166 million hectares) and grazed forest (23 million hectares) that comprise 188 million hectares of the nation's nonfederal rural land. It emphasizes conservation practices that are routinely applied on rangelands west of the 100th meridian to accomplish multiple management and environmental goals, including maintenance of plant community health, protection of water quality and quantity, reduction of accelerated soil erosion, and promotion of economic stability through rangeland sustainability. Conservation practices are usually not implemented in isolation, but as part of a broader conservation plan that may potentially recommend implementation of multiple practices. Resource management systems represent a combination of conservation practices and resource management actions prescribed to address multiple natural resource concerns that meet or exceed the quality criteria for resource sustainability. It is fully anticipated that some combination of agricultural and environmental benefits arise from implementation of these conservation practices on rangelands, but quantitative measures of their specific effects on soil, water, animals, plants, and air are required to document the efficacy of these practices and systems.

Organization and Approach

Rangeland CEAP encompasses four interrelated components:

1. *National Assessment*. Evaluation of the effects of conservation management on rangelands across the United States accomplished with a combination of ground-based measurements, remotely sensed data, and hydro-ecological and economic simulation models. This effort is coordinated by the ARS with emphasis on watershed modeling in the Intermountain West (Weltz et al. 2008).
2. *Watershed Assessment Studies*. Quantification of the measureable cumulative effects of conservation practices and enhancement of understanding of the interactions among practices in experimental watersheds. These watersheds occur in both croplands and grazing lands and are intended to provide in-depth assessments that are not possible at the regional scale to evaluate and enhance performance of the national assessment models.
3. *Bibliographies*. Compilation of published literature citations addressing the environmental benefits of conservation practices and programs for grazing lands was completed by the USDA National Agricultural Library in 2006. Dynamic bibliographies using real-time searches in the National Agricultural Library catalog (AGRICOLA) have been assembled (USDA National Agricultural Library 2007).
4. *Literature Synthesis*. Compilation of the current status of knowledge concerning the ecological effectiveness of major conservation practices applied on rangelands by systematically mining the published scientific literature.

Rangeland Synthesis

Rangelands synthesis CEAP has been developed to provide an in-depth assessment of the published experimental information concerning the effectiveness of previously implemented conservation

practices on rangelands. The primary goal is to provide the most definitive assessment of conservation impacts ever conducted within the rangeland profession to serve as an evidence-based benchmark for the efficacy of current conservation practices. This is a necessary and essential step for assessing the benefits of existing conservation practices and determining whether or not current practices require modification in either design or implementation to enhance their effectiveness in future programs. This information, coupled with evidence-based recommendations to enhance conservation programs, and identification of key knowledge gaps in existing information will promote development of novel evidence-based conservation systems that possess the capacity to assess environmental quality and ecosystem services in addition to traditional agricultural production metrics.

The rangeland literature synthesis was specifically organized around a series of testable questions derived from the stated purposes or outcomes of seven major conservation practices as identified in the NRCS National Conservation Practice Standards. These conservation practices were selected for assessment based on their prominence in the conservation planning environment, the extent and frequency with which they are applied, and the amount of incentive payments allocated to them (Tables 1 and 2). Rigorous literature syntheses established the portion of experimental studies that supported, refuted, or were insufficient to assess the benefits of these conservation practices. Two

TABLE 2. Environmental Quality Incentive Program funds (US dollars) expended on the seven major conservation practices address in the rangeland synthesis by region and state during 1997–2003.

Region	State	Brush management	Prescribed burning	Prescribed grazing	Range planting	Riparian herbaceous cover	Upland wildlife habitat management
West rangeland state							
	Arizona	672 345		1 090 536	134 842		1 650
	California	2 192 285	2 625	110 813	592 108	12 116	20 394
	Colorado	185 295	3 785	3 496 391	70 985		8 936
	Idaho	7 250		29 411	18 868	1 126	527
	Montana			923 457	135 236	148	15 895
	Nevada	35 756	90	770	77 883		214
	New Mexico	3 259 774	21 542	421 262	674 895		188
	Oregon	164 759		83 377	188 088	66	35 790
	Utah	415 038	13 402	18 199	241 036	5 629	1 113
	Washington	563		41 862	96 686		75 547
	Wyoming	145 829	5 145	3 136			108
West total		7 078 894	46 589	6 219 214	2 230 627	19 085	160 362
Central rangeland state							
	Kansas	551 470	33 595	1 321 533	142 215		12
	Nebraska	124 609	2 218	197 443	133 641		21 757
	North Dakota	8 807		1 091 930	203 208		81 976
	Oklahoma	2 215 107	200 046	298 249	174 530		430
	South Dakota	6 376		59 330	100 057		41 820
	Texas	9 297 443	72 136	122 491	1 535 268	25 682	5 290
Central total		12 203 812	307 995	3 090 976	2 288 919	25 682	151 285
Grand total		19 282 706	354 584	9 310 190	4 519 546	44 767	311 647

“The recent success of CEAP provides numerous opportunities and challenges to achieve its full potential within the USDA and the broader conservation community.”

additional chapters emphasizing landscape ecology and socioeconomic issues, including ecosystem services, were developed on the basis of their anticipated importance to future conservation programs and planning. These final two chapters were organized to be cross-cutting among all seven conservation practice standards.

Seven major conservation practices and two cross-cutting issues are addressed in the Rangeland CEAP literature synthesis.

- Prescribed Grazing
- Prescribed Burning
- Brush Management
- Range Planting
- Riparian Herbaceous Cover
- Upland Wildlife Habitat Management
- Herbaceous Weed Control
- Landscape Analysis (cross-cutting chapter)
- Socioeconomics and Ecosystem Services (cross-cutting chapter)

A writing team was formed for each of these nine chapters by recruiting team leaders with recognized experience and expertise in the respective subject matter areas and encouraging them to select two to four subject matter specialists with sufficient diversity to address the entire scope of ecological topics under consideration—soils, water, air, plants, and animals—as they relate to the seven major conservation practices. Geographic representation of team members across US rangelands was considered in the selection process to the extent possible. Teams focused on the development of tabular databases comprising quantitative information addressing multiple ecological responses to conservation practices to provide an unprecedented compilation of evidence-based information. Databases were primarily derived from the refereed literature with some quality “grey” literature included at the discretion of the writing teams. Individual chapters underwent rigorous peer review by three recognized experts that were not affiliated with CEAP; reviewer recommendations were provided to the chapter authors for incorporation, and the revised chapters were evaluated by the academic coordinator of Rangeland CEAP. The entire document was evaluated for relevance and impact by one nonfederal reviewer and one NRCS reviewer prior to publication.

Major sections addressed within each of the synthesis chapters include the following:

- Description of conservation practices and their purported benefits.
- Evidence-based assessment of conservation benefits, including potential tradeoffs and risks of not implementing the practice, and of unintended negative outcomes.
- Recommendations to modify or develop alternative conservation practices to more effectively accomplish the intended purposes.
- Identification of critical knowledge gaps in current information.
- Succinct summary and conclusion of findings for each conservation practice.
- Literature-cited section containing citations within the text, but not those used to support the extensive tabular data. These supporting citations will be made available in a searchable electronic version of this document.

The rangeland literature synthesis is available in both hardcopy and electronic formats. The electronic version will be posted on the NRCS-CEAP, National Agricultural Library, and Society for Range Management Web sites and it will be searchable for both citations and appendices of tabular data specific to each chapter. This document is designed to target multiple audiences, including 1) policy makers (e.g., executive summary), 2) practitioners and students (e.g., general synthesis), and 3) researchers and modelers (e.g., tabular databases and supporting references).

CEAP IMPLEMENTATION AND THE ROAD AHEAD

Design and implementation of conservation practices through use of the best available information and technology is a hallmark of NRCS. The knowledge generated through CEAP-sponsored assessments is critical to continuation of this mission by optimizing the cost-effectiveness of conservation practices and the environmental outcomes

CEAP has created new opportunities to promote rangeland stewardship. (Photo: Rick Miller)

that they support. CEAP has generated new conservation opportunities to manage agricultural landscapes for environmental quality, created diverse and valuable conservation partnerships, and emphasized conservation assessment and planning at the watershed and landscape scales.

Anticipated applications of the information created by CEAP include the following:

- Support further development of grazing lands management and conservation practices within the Soil and Water Resources Conservation Act and National Conservation Program.
- Informing grazing land initiatives in subsequent Farm Bills.
- Advancement of conservation planning tools and program delivery mechanisms for targeted implementation and enhanced adoption.
- Evaluation of mitigation and adaptation strategies associated with climate change, water security challenges, or changes in land use or management.
- Devising inventory and monitoring protocols to better document conservation benefits for both agricultural production and environmental quality.

The recent success of CEAP provides numerous opportunities and challenges to achieve its full potential within the USDA and the broader conservation community. Implementation of CEAP will require reevaluation of procedures concerning conservation planning, greater knowledge transfer among USDA programs, modification of select conservation practices, and additional technology development and transfer. An expanded culture of collaboration among USDA programs and agencies, and several nonfederal partners, has contributed greatly to the transformational influence of CEAP. Continued collaboration is necessary both within USDA programs as well as with the broader conservation and agricultural communities to further capitalize on the knowledge and unprecedented capacity associated with rapidly emerging conservation science to produce the next generation of conservation programs for the 21st century.

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CHAPTER

1

An Evidence-Based Assessment of Prescribed Grazing Practices

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“Experimental grazing research has produced consistent relationships between stocking rate and plant production, animal production, and species composition of herbaceous plant communities.”

An Evidence-Based Assessment of Prescribed Grazing Practices

1

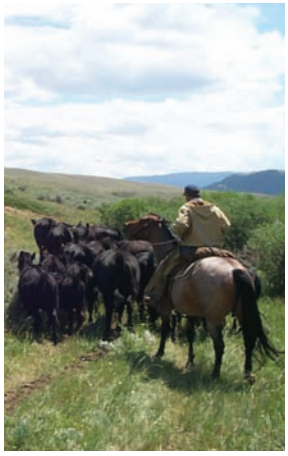
David D. Briske, Justin D. Derner, Daniel G. Milchunas, and Ken W. Tate

INTRODUCTION

Prescribed grazing is inclusive of many interrelated management and conservation activities implemented for purposes of managing grazed ecosystems. It is supported by a loosely organized information base that contains management experience, agency policy and procedures, and scientific information that has been developed throughout the history of the rangeland profession. The components of prescribed grazing are implemented in various combinations to achieve multiple management goals and outcomes under a wide variety of ecological conditions in diverse rangeland ecosystems. A fundamental premise of effective grazing management is that it supports ecosystem sustainability and restoration of degraded ecosystems. Management actions have traditionally emphasized livestock species and number and their temporal and spatial distribution on the landscape (Stoddart et al. 1975; Heitschmidt and Taylor 1991). The management of grazed ecosystems involves multiple human dimensions as well as complex ecological processes, making it difficult and impractical to attempt to separate grazing management from overall enterprise management (Stuth 1991). Therefore, management practices are commonly designed and applied within the context of specific landowner operations, management needs, and natural resource conservation goals. Consequently, prescribed grazing involves a continuum of management activities ranging from extensive management to those that are much more labor and infrastructure intensive.

Context for the initial development of prescribed grazing in the United States originated with management recommendations to promote sustainable use and recovery of rangelands damaged by excessive livestock grazing early in the 20th century (Smith 1896; Wooten 1916; Sampson 1923, 1951; Hart and Norton 1988). Excessively high stocking rates (animal units area^{-1} time^{-1}) common to the late 19th and early 20th centuries were unsustainable, and the negative consequences of those extreme stocking rates adversely affected numerous ecosystems throughout the Great Plains and West. Early rangeland research advocated use of reduced stocking rates and simple grazing systems to impose early season deferment and season-long rest to halt and potentially reverse ecological damage created by severe overgrazing. Increased efficiency of livestock production became an important objective during the 1980s and was associated with the introduction of short-duration grazing to the United States (Savory and Parsons 1980; Savory 1983). These management systems were designed to improve the efficiency of forage harvest, enhance forage quality, and promote livestock production. More recently, prescribed grazing has emphasized broader conservation goals and ecosystem services. Biodiversity conservation, water quality and quantity, woodland encroachment, invasive species, and carbon sequestration are but a few of the current high-profile conservation issues considered within grazed ecosystems. However, this emphasis is rather recent, and the amount of experimental information to date is insufficient to draw valid generalizations regarding these conservation issues.

Even though the primary objective of this body of information is to promote effective grazing management, this is in itself not a sufficient foundation on which to evaluate this important land use. It is essential that the underlying components and processes of effective grazing management be recognized, understood, and documented to ensure that this information base is carefully scrutinized, accurate, and



Prescribed grazing is the management of vegetation harvest by grazing or browsing animals to achieve desired objectives. (Photo: Derek Bailey)

effectively promoted. The primary objective of Rangeland CEAP is to organize and evaluate the current body of scientific information supporting the anticipated benefits of rangeland conservation practices implemented by the US Department of Agriculture, Natural Resources Conservation Service (USDA-NRCS). This assessment is intended to provide the foundation for the next generation of planning and assessment procedures that are to emphasize environmental quality and the assessment of multiple ecosystem services in addition to the traditional outcomes of farm and ranch productivity (Maresch et al. 2008).

This chapter evaluates the ecological effectiveness of the major purposes and purported benefits for prescribed grazing as described in the USDA-NRCS National Conservation Practice Guidelines. This standard defines prescribed grazing as managing the harvest of vegetation with grazing and/or browsing animals that is often applied as one component of a broader conservation management system to achieve one or more of the following purposes:

- Improve or maintain desired species composition and vigor of plant communities.
- Improve or maintain quantity and quality of forage for grazing and browsing animals' health and productivity.
- Improve or maintain surface and/or subsurface water quality and quantity.
- Improve or maintain riparian and watershed function.
- Reduce accelerated soil erosion and maintain or improve soil condition.
- Improve or maintain the quantity and quality of food and/or cover available for wildlife.
- Manage fine fuel loads to achieve desired conditions.

This definition is very similar to that provided in the Society for Range Management (SRM) Glossary of Terms (1998)—"the manipulation of animal grazing in pursuit of a defined objective"—and to that of targeted grazing—"the application of a specific kind of livestock at a determined season, duration and intensity to accomplish a defined objective" (Launchbaugh and Walker 2006). Targeted grazing emphasizes objectives associated with landscape dynamics in addition to livestock production. It is important to note that prescribed grazing, as defined above, is a much broader concept than grazing system, which describes a specialized application of grazing management based on recurring periods of grazing, rest, and deferment for two or more pastures (Heitschmidt and Taylor 1991). The NRCS National Range and Pasture Handbook describes prescribed grazing schedules to recommend appropriate periods of grazing, rest, and deferment (USDA-NRCS 2003).

The experimental data addressing these purposes were extracted primarily from the peer-reviewed literature, summarized and incorporated into tabular forms to provide an evidence-based assessment of how well prescribed grazing achieves these stated purposes. In some instances, direct comparisons could be made between intended conservation outcomes and the experimental evidence, but in many others, inferences had to be drawn from the most relevant experimental data to assess the effectiveness of conservation outcomes. Constraints of experimental research have influenced both the type of information available and the investigations selected for inclusion in this assessment. For example, spatial heterogeneity may produce conditions where most pastures under consideration possess generally similar topoedaphic characteristics, but in other cases one or more pastures may possess distinctly different characteristics. Only the first condition characterized by relatively homogeneous site conditions meets the traditional experimental requirements of replication and comparison with experimental controls, while decisions regarding heterogeneous site conditions can be assessed only on a case-by-case basis. Given that the goal of this chapter was to evaluate the preponderance of evidence supporting major grazing management practices, investigations that were unreplicated or that did not have experimental controls, that applied unequal treatments, or that contained minimal data were not included. These requirements were relaxed to some extent for the evaluation of wildlife because investigations addressing responses of specific wildlife species or groups to unique management practices were often limited. Similarly, minor wildlife groups were

not addressed because of limited research evidence and space limitations.

This chapter is organized into six major headings: introduction, evaluation of prescribed grazing purposes, associated considerations, recommendations, knowledge gaps, and conclusions. The evaluation of prescribed grazing purposes is the largest section, and it contains seven secondary headings addressing each of the conservation purposes previously described. Several of these purposes are further subdivided into tertiary headings of stocking rate and grazing system because these two research themes contain a large portion of the experimental information associated with grazing management. In addition to summarizing the experimental evidence relevant to prescribed grazing management, this chapter emphasizes the strengths and weaknesses of the experimental data, provides recommendations for improvement of this conservation practice, and identifies major knowledge gaps in the experimental literature. The overarching goal is to describe the current status of grazing management information to provide a foundation for development of the next generation of prescribed grazing practices. This chapter was commissioned by and is directed toward the NRCS, but it also contains important implications to the broader rangeland profession.

EVALUATION OF PRESCRIBED GRAZING PURPOSES

Improve or Maintain Desired Species Composition and Vigor of Plant Communities

Stocking Rate. Stocking rate has long been recognized as a fundamental variable determining the sustainability and profitability of grazed rangeland ecosystems (Smith 1896; Wooton 1916; Sampson 1923). The objective of stocking rate is to balance the forage demand of grazing animals with that of forage production over an annual forage production cycle. The difficulty encountered when setting and maintaining appropriate stocking rate on rangelands is the high variability of forage production associated with annual and interannual precipitation variation. It is often recommended that stocking rates should be conservatively applied to minimize the detrimental consequences of overstocking during drought on the economic and ecological sustainability of grazed ecosystems.

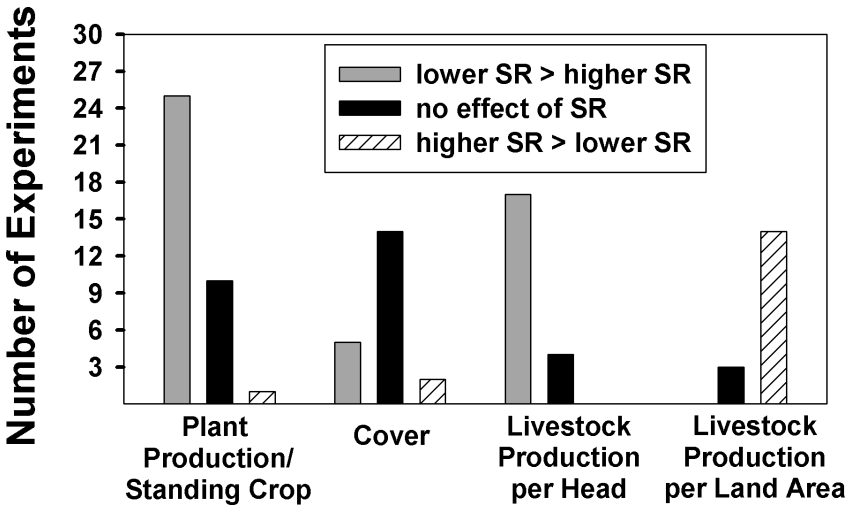


FIGURE 1. Number of investigations reporting significant effects of stocking rate on plant production and cover and livestock production per head and per unit land area.

The importance of stocking rate to the management of grazed ecosystems has attracted considerable research attention over the past several decades. This research has produced consistent relationships between stocking rate and plant production, animal production, and species composition of herbaceous plant communities. Plant production decreases with increasing stocking rate, as does individual animal production (Bement 1969; Manley et al. 1997; Derner and Hart 2007; Derner et al. 2008). In contrast, animal production per land area increases with increasing stocking rate within the limits of ecosystem sustainability. These ecosystem responses to stocking rate have clear production and conservation implications.

The response of several ecosystem variables indicates that stocking rate is at least indirectly correlated with ecosystem function and sustainability. High grazing intensities generally appear to minimize ecosystem function, which often has negative consequences for conservation goals and the provisioning of ecosystem services. Plant production is the most consistent response with 69% (25 of 36) of the investigations reporting greater plant production at lower compared to higher stocking rates (Fig. 1). Twenty-eight percent (10 of 36) showed no difference in plant production with stocking rate. Only four of

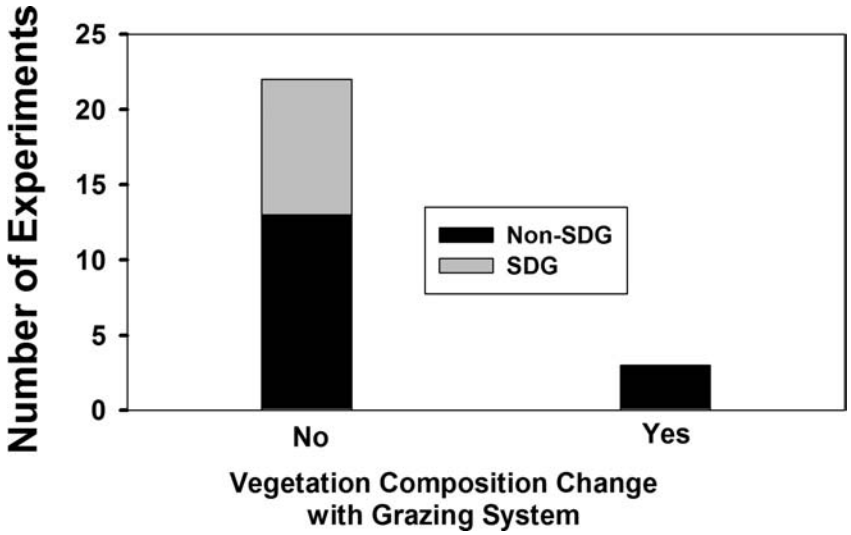


FIGURE 2. Number of investigations reporting significant effects of grazing system, categorized as short-duration and non–short-duration systems, on favorable changes in species composition of plant communities.

these investigations considered plant species diversity or richness in relation to stocking rate, but the trend is for increasing diversity and richness with increasing stocking rate, which is a consistently observed community response to grazing (Milchunas et al. 1988). This response is interpreted as a function of the suppression of grass dominants at high stocking rates, which increases resource availability for subordinate species within the community (Collins 1987; Anderson and Briske 1995; Knapp et al. 1999). However, cases do exist where intensively grazed ecosystems are required to provide specific habitat for flora and fauna (Milchunas and Lauenroth 2008; Derner et al. 2009).

Stocking rate has tremendous potential to modify the species composition of herbaceous vegetation. Significant change in species composition was documented to occur in 71% (17 of 24) of the stocking rate studies evaluated. Plant cover showed a much less consistent response than did either production or species composition with 67% (14 of 21) of the investigations showing no difference with stocking rate compared to 24% (5 of 21) that did show a positive response. Compositional changes largely follow the classical increaser–decreaser patterns outlined by Dyksterhuis (1949) and more recently verified in a global vegetation analysis (Diaz et al. 2007) in which tallgrasses are replaced by midgrasses and midgrasses by shortgrasses. Eight of these studies recorded vegetation responses for ≥ 20 yr and 14 studies for ≥ 10 yr, but significant vegetation change was also recorded in shorter time periods. These vegetation responses also document the occurrence of equilibrium dynamics in which grazing modifies the species composition of plant communities in addition to weather conditions (Fuhlendorf et al. 2001; Briske et al. 2003). The potential for recovery of species composition in response to reduced stocking rates also documents the high degree of resilience associated with many rangeland ecosystems (Milchunas et al. 1988).

Grazing Systems. Although changes in plant species composition are often more qualitatively assessed than the plant and animal production values presented previously, the majority of investigations have not shown a clear benefit of rotational grazing over continuous grazing in promoting secondary succession or improving community composition on rangelands (Holechek et al. 1999, 2006). In our survey of 25 grazing experiments, 86% (18) indicated no difference in species composition for continuous compared to rotational grazing at comparable stocking rates. Only 3 of 25 experiments recorded improvements in species composition, and these were all deferred-rotation rather than short-duration systems (Fig. 2).

Experimental data referencing biotic diversity in grazing systems are limited, especially at regional scales, so definitive conclusions are unattainable at this point. However, the limited experiments addressing plant species diversity do not show that grazing systems enhance plant species diversity (Holechek et al. 2006). In tallgrass prairie, grazing system did not influence plant richness or diversity, but both variables increased with increasing stocking rate (Hickman et al. 2004). Increasing stocking rate reduced the abundance of the several dominant C_4 grass species and increased the expression of several subordinate species. Plant diversity responses to grazing are dependent on the direct response of various species to grazing and the indirect response of other species to grazing-induced release from competition (Milchunas et al. 1988; Anderson and Briske 1995).

Grazing Season and Deferment. Research addressing the season and length of grazing deferment is surprisingly limited given its

importance to grazing management. It is difficult to draw inferences from the few investigations specifically addressing season of grazing, especially given the variability in production and defoliation responses associated with precipitation variation within and among years (Zhang and Romo 1994). These authors were unable to make conclusive recommendations regarding production responses of northern mixed prairie to the seasonality and frequency of defoliation because of weather variation between years. Plant species with unique growth periods and production potentials contribute additional complexity to this assessment (Volesky et al. 2004). This underscores the difficulty of making generalizations regarding the appropriate season of grazing and deferment.

Inferences regarding the appropriate length of grazing deferment can be derived from grazing systems research previously evaluated. Short deferment periods do not yield benefits in those variables measured, that is, plant and animal production, species composition, and soil characteristics. It can be inferred from this extensive data set that successive short deferments of 30–45 d are ineffective in offsetting short, intensive grazing periods of 2–11 d. Conclusions regarding length of deferment have been drawn from comparisons of short-duration and high-intensity, low-frequency systems using 42- and 84-d deferment periods, respectively (Taylor et al. 1993). These authors concluded that 80–90-d deferment periods were required to maintain desired species composition on semiarid rangelands. This interpretation has been corroborated by research conducted in mesic tallgrass ecosystems (Reece et al. 1996). Specific ecological mechanisms limiting increased plant production and improved species composition in response to short-term periodic deferment in rotational systems are not entirely clear, but they are very likely influenced by the time required for plant recovery, especially on semiarid rangelands, and the coincidence of favorable growth conditions with periods of grazing deferment (Briske et al. 2008).

Grazing deferment relative to the onset and recovery from drought has also received minimal attention given its significance to grazing management. However, several conclusions can be drawn from a valuable, but limited data set. First, grazing deferment during drought has minimal potential to enhance plant production or species composition, even though it is often necessary to destock because of insufficient forage availability (Eneboe et al. 2002; Heitschmidt et al. 2005; Gillen and Sims 2006). However, deferment is important to maintain sufficient plant cover and density to protect soil quality and promote plant recovery once rainfall resumes (Wood and Blackburn 1981a&b; Thurow 1991; Dalgleish and Hartnett 2006). Second, grazing deferment is not necessarily required for rapid and effective vegetation recovery from moderate drought conditions (Eneboe et al. 2002; Heitschmidt et al. 2005). Investigations demonstrating the ability of rainfall to override the effects of stocking rate on forage production and species composition indirectly support this interpretation (Milchunas et al. 1994; Biondini et al. 1998; Gillen et al. 2000; Vermeire et al. 2008). Third, in the cases involving severe, prolonged drought, 2 yr or more may be required for recovery of species composition and productivity. Severe, multiyear drought can induce mortality of plants and tillers to retard plant growth following the resumption of rainfall (Briske and Hendrickson 1998; Dalgleish and Hartnett 2006; Yahdjian et al. 2006). Consequently, several growing seasons may to be required for tiller and plant densities to recover to predrought values. Plant mortality was found to be approximately twice as great in heavily compared to more lightly grazed Great Plains rangelands following the multiyear drought of the 1950s (Albertson et al. 1957). Greater plant mortality is likely a consequence of the suppressed root growth and function that is known to occur with severe grazing of individual plants (Crider 1955).

> “Research addressing the season and length of grazing deferment is surprisingly limited given its importance to grazing management.”

Improve or Maintain Quantity and Quality of Forage for Grazing and Browsing Animals' Health and Productivity

Stocking Rate. Experimental data confirm the occurrence of a consistent trade-off between animal production per head and per land area with increasing stocking rate. Eighty percent (16 of 20) of investigations reported greater animal production per head at low compared to high stocking rates, while 82% (14 of 17)

“Individual plant production is most greatly suppressed by defoliation during the middle of the growing season, which coincides with culm elongation and the early boot stage of inflorescence development…”

showed greater animal production per land area at high compared to low stocking rates (Fig. 1). This trade-off in animal performance is readily explained by the greater availability of plant biomass per individual animal with decreasing stocking rate and greater forage harvest per unit land area with increasing stocking rate. Experimental evidence indicates that both forage quantity and quality decrease with increasing high compared to low stocking rates. Forage quality is most likely to decrease during grazing periods insufficient for appreciable regrowth, where animals initially select the highest-quality forage.

Pattern of tiller defoliation research shows that 80% or more of all tillers can be defoliated in a single grazing period with high stocking rates or grazing intensities (8 of 11 studies). This indicates that high harvest efficiencies and uniform grazing patterns can be obtained with large livestock numbers in certain cases. However, these data also indicate that multiple defoliations occur early within a grazing cycle (four of six studies). It has been suggested that repeat defoliations begin to occur at about the time that 60% of the tillers are initially defoliated and that very high grazing pressures and paddock numbers would be required to minimize the occurrence of multiple grazing events within individual grazing periods (Jensen et al. 1990a). These data challenge the widely held assumption that rotational grazing restricts grazing to a single event per plant during short (5–10-d) grazing periods while simultaneously promoting high plant utilization. This may indicate why minimal differences in plant defoliation patterns have been found between rotational and continuous grazing (Hart et al. 1993b). These detailed investigations were conducted on very small pasture sizes (0.2–24 ha), so caution is warranted when scaling these responses to larger areas. In addition, frequency of defoliation may be more detrimental to plants when defoliation events are separated by periods of regrowth, as indicated below.

Frequency of tiller defoliation consistently increases with increasing stocking rate (9 of 10 studies). Defoliation intensity of individual tillers also increases with increasing stocking rate, but not as rapidly as defoliation frequency (four of five studies).

Forage quality decreased with increasing stocking rate within an individual grazing period in all four studies evaluated and with increasing time of grazing for all three studies that carefully evaluated this relationship. This clearly indicates that animals compete for quality forage, and this process establishes the basis for the negative response of individual animal performance with increasing stocking rate.

Season of plant defoliation has unique and consistent effects on plant production. Individual plant production is most greatly suppressed by defoliation during the middle of the growing season, which coincides with culm elongation and the early boot stage of inflorescence development, especially in bunchgrasses (Olson and Richards 1988). This was documented in six of nine investigations and in all three studies specifically evaluating growth stage responses to defoliation. Early season defoliation had the least detrimental effect on subsequent plant production, and late season defoliation had an intermediate effect. However, plant production is increasingly suppressed with increasing frequency and intensity of defoliation at any stage of growth (five of six studies), confirming the interpretation that multiple defoliations within a growing season are detrimental to plant growth and function (Reece et al. 1996; Volesky et al. 2004). These patterns of grass production responses to defoliation at various phenological stages substantiate the criticism that has been directed toward early season deferment (i.e., range readiness) as a valid conservation practice.

The occurrence of patch grazing has been well documented in several investigations, and it appears to directly relate to the nutritional intake of animals when other constraints on animal distribution are absent (e.g., distance to water and topography). Previously grazed patches support forage of higher nutritional quality, including crude protein, fiber, and digestibility, even though forage quantity may be less than on previously ungrazed patches (Cid and Brizuela 1998; Ganskopp and Bohnert 2006). The primary mechanism contributing to patch grazing is animal aversion to consumption of senescent plant

material, especially current and previous year's culms or stems (Ganskopp et al. 1992, 1993). Consequently, patch grazing may provide a nutritional benefit to animals at low and moderate stocking rates (Cid and Brizuela 1998).

Patch structure is relatively consistent within season and among years, but it is less stable at higher than at lower stocking rates (Willms et al. 1988; Cid and Brizuela 1998). At higher stocking rates, animals begin to selectively graze previously ungrazed patches to maintain sufficient forage intake, and they forage greater distances to achieve this goal (Ring et al. 1985; Ganskopp and Bohnert 2006). Patch grazing can be minimized by the removal of senescent biomass, especially previous year's biomass with fire, mowing, or periodic heavy stocking (Ganskopp and Bohnert 2006). However, the implications of patch grazing have been shifting from that of an inefficient use of forage by livestock to a desirable component of vegetation heterogeneity capable of promoting biodiversity in the Great Plains (Fuhlendorf and Engle 2001, 2004). This is an especially relevant consideration, both within and among pastures, in light of the CEAP initiative, which emphasizes management for environmental quality and multiple ecosystem services as well as production goals.

Grazing Systems. Grazing systems represent a specialization of grazing management that defines the periods of grazing and nongrazing (Heitschmidt and Taylor 1991; SRM 1998), and they have been given tremendous emphasis by both managers and researchers. It is important to recognize that constraints of experimental research, including the need for relatively homogeneous site conditions necessary for replication and comparison with experimental controls, has emphasized the potential for various periods of grazing and rest to alter the ecological processes controlling plant and animal production. They are unable to—and therefore do not—address livestock distribution in heterogeneous landscapes or livestock movement in response to site readiness along elevation gradients. However, these latter considerations are also important and have been addressed with experimental data collected with more appropriate experimental approaches.

The major experimental investigations of grazing systems have been categorized by geographic location, ecosystem type, relative stocking rate, and number and size of pastures for each of the respective investigations (Briske et al. 2008). Variables were indicated to differ between continuous and rotational grazing only when they were reported as being statistically significant by the authors. For each experiment, plant and/or animal production (the most quantitative data collected) was characterized as 1) greater for continuous grazing (CG > RG), 2) greater for rotational grazing (RG > CG), or 3) equal if differences did not exist between continuous and rotational grazing (ND). These comparative responses were summarized and presented as separate histograms for those investigations that used similar stocking rates between grazing treatments (Fig. 3A), those that used greater stocking rates for rotational than for continuous grazing (Fig. 3B), and for all stocking rates combined (Fig. 3C). These experimental comparisons of rotational systems included five studies conducted for 9 yr or more, and four had pasture sizes greater than 300 ha, but only two had greater than eight pastures per grazing system.

Eighty-nine percent of the experiments (17 of 19; Appendix I) reported no differences for plant production/standing crop between rotational and continuous grazing with similar stocking rates (Fig. 3A). When stocking rate was less for continuous than rotational grazing, 75% of the experiments (three of four) reported either no differences or greater plant production for continuous grazing (Fig. 3B). Across all stocking rates, 83% of the experiments (19 of 23; Appendix I) reported no differences for plant production between rotational and continuous grazing, 13% (three) reported greater plant production for rotational compared to continuous grazing, and 4% (one) reported greater production for continuous grazing (Fig. 3C; Briske et al. 2008).

Fifty-seven percent of the experiments (16 of 28; Appendix I) reported no differences for animal production per head between rotational and continuous grazing with similar stocking rates, and 36% (10) reported greater per head production for continuous grazing (Fig. 3A). When stocking rate was less for continuous than rotational grazing,

Grazing is a major land use on 188 million hectares of non-federal lands in the Great Plains and western U.S. (Photo: Sonja Smith)

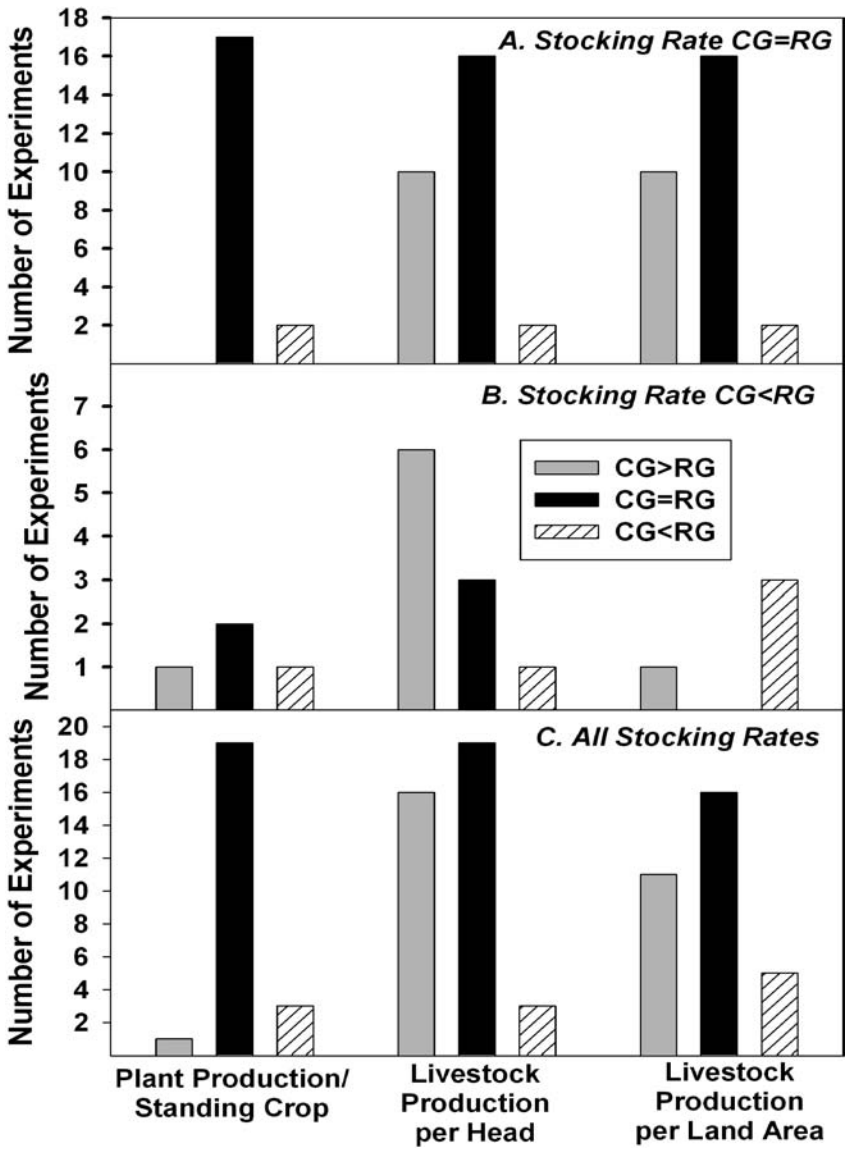


FIGURE 3. Number of published grazing experiments that reported significantly higher, equal, or lower plant and animal production responses for continuous compared to rotational grazing at (**A**) similar stocking rates, (**B**) higher stocking rates for rotational grazing, and (**C**) across stocking rates for all experiments. Animal production is presented as both a per head and a per land area response (from Briske et al. 2008).

90% of the experiments (9 of 10) reported either similar or greater per head animal production for continuous grazing (Fig. 3B). Across all stocking rates, 50% (19 of 38; Appendix I) of the experiments reported no differences for animal production per head between rotational and continuous grazing, 8% (three) reported greater production for rotational grazing, and 42% (16) reported greater production for continuous grazing (Fig. 3C). Fifty-seven percent of the experiments (16 of 28; Appendix I) reported no differences for animal production per unit land area between rotational and continuous grazing with similar stocking rates, and 36% (10) reported advantages for continuous grazing (Fig. 3A). When stocking rate was lower for continuous than rotational grazing, 75% (three of four; Appendix I) of the experiments reported greater animal production per area for rotational grazing (Fig. 3B). Across all stocking rates, 50% (16 of 32; Appendix I) of the experiments reported no differences for animal production per land area between rotational and continuous grazing, 16% (five) reported greater production for rotational grazing, and 34% (11) reported greater production for continuous grazing (Fig. 3C; Briske et al. 2008). A recent ranch-scale investigation comparing four grazing systems over a 7-yr period that was not included in this numerical assessment also reported minimal differences in livestock production among grazing systems (Pinchak et al. 2010).

No evidence was found indicating that grazing systems override livestock preference for site selectivity. Comparisons of continuous season-long and rotational grazing on five range sites in northern mixed-grass prairie found no differences among grass utilization over a 2-yr period (Kirby et al. 1986). This occurred in spite of the fact that the rotational system had both a higher stocking rate and a higher stock density than did the continuous system. Heitschmidt et al. (1989) corroborated these conclusions in mixed-grass prairie in north-central Texas. Paddocks of 30 and 10 ha were used to simulate rotational grazing systems with 14 and 42 paddocks. Livestock selectivity was not modified by either rotational grazing system compared to continuous grazing. These authors concluded that forage availability, rather than stocking density or grazing system, was the primary mechanism that modifies animal selectivity. However, none of these investigations specifically addressed the presence of riparian systems in which livestock frequently congregate (George et al., this volume).

Only four studies were found that directly compared forage quality in rotational and continuous grazing. Forage quality was

comparable among systems in two of the investigations (Jung et al. 1985; Heitschmidt et al. 1987b), and one each favored continuous (Pfister et al. 1984) and rotational grazing (Heitschmidt et al. 1987a). Forage quality was greater for a seven-pasture short-duration system compared to a seven-pasture high-intensity, low-frequency system, but similar to that of a Merrill four-pasture, three-herd system on the Edwards Plateau of Texas (Taylor et al. 1980). Tiller defoliation patterns in continuous and rotational grazing have received only minimal attention, but frequency and intensity of tiller defoliation was greater for rotational grazing in only one (Senock et al. 1993) of four investigations (Hart et al. 1993a; Derner et al. 1994; Volesky 1994). Collectively, the small number of investigations reporting mixed results makes conclusions regarding grazing systems effects on forage quality and defoliation patterns equivocal compared to conclusions addressing plant and animal production, and species composition.

Three categories of evidence exist to explain why intensive rotational grazing systems have not shown greater quantity and quality of forage and animal production in experimental research. First, short, periodic deferments based on established schedules do not always coincide with favorable growth conditions in rangeland environments (e.g., Taylor et al. 1993; Holechek et al. 2001; Gillen and Sims 2006). The amount and variability of rainfall and the associated predictability, duration, and amount of plant growth appear to override the potential benefit derived from the redistribution of grazing pressure in space and time in rotational grazing systems (O'Reagain and Turner 1992; Ash and Stafford Smith 1996; Holechek et al. 2001; Ward et al. 2004). Plant growth and improvement in species composition will be promoted primarily when deferment coincides with environmental conditions favorable for plant growth (Heitschmidt et al. 2005; Gillen and Sims 2006).

Improper grazing can detrimentally affect soil surface characteristics to accelerate runoff and erosion. (Photo: Ken Tate)

“These results refute prior claims that animal trampling associated with high stocking rates or grazing pressures in rotational grazing systems enhance soil properties and promote hydrological function”

Second, rotational grazing may not control the frequency and intensity of plant defoliation as effectively as often assumed (Gammon and Roberts 1978a, 1978b, 1978c; Hart et al. 1993a). Investigations of tiller grazing patterns indicate that it is difficult to achieve a high percentage of tiller defoliation (> 80%) before multiple defoliations begin to occur within a single grazing period (Jensen et al. 1990a; O'Reagain and Grau 1995). These data indicate that grazing management strategies only marginally modify animal selectivity within the range of conditions that have been evaluated. Third, forage quality is not consistently or substantially increased in intensive systems compared to continuous grazing (Denny et al. 1977; Walker et al. 1989; Holechek et al. 2000). The absence of experimental evidence supporting these three major underlying assumptions associated with rotational systems is consistent with the production responses generated from experimental comparisons of rotational and continuous grazing. However, conclusions addressing tiller defoliation patterns are derived from a small number of experiments conducted in very small pastures (0.2–24 ha) that may not be entirely representative of grazing patterns at larger scales.

These experimental results collectively indicate that rotational grazing does not promote primary or secondary production compared to continuous grazing within rangeland ecosystems. These interpretations are consistent with those of previous reviews over the past 50 yr (Heady 1961; Van Poollen and Lacey 1979; Holechek et al. 2001), and they clearly support the long-standing conclusion that stocking rate and weather variation account for the majority of variability associated with plant and animal production on rangelands (Van Poollen and Lacey 1979; Heitschmidt and Taylor 1991; Gillen et al. 1998; Holechek et al. 2001; Derner and Hart 2007).

Improve or Maintain Surface and/or Subsurface Water Quality and Quantity

Stocking Rate. The response of soil hydrological characteristics to grazing largely parallel those of other ecological variables because stocking rate is the most important driver regardless of grazing system (Wood and Blackburn 1981a&b; Thurow 1991). This occurs because the removal of large amounts of plant cover and biomass by intensive grazing reduces the potential to dissipate the energy of raindrop impact and overland flow. The erosive energy of water and the long-term reduction of organic matter additions to soil detrimentally affect numerous soil properties, including the increase of bulk density, disruption of biotic crusts, reduced aggregate stability, and organic matter content, which collectively reduce infiltration rate and increase sediment yield and runoff. Animal trampling is another source of mechanical energy that breaks soil aggregates and is therefore negatively correlated with maintenance of soil structure necessary for high infiltration rates (Warren et al. 1986b; Thurow 1991; Holechek et al. 2000). These results refute prior claims that animal trampling associated with high stocking rates or grazing pressures in rotational grazing systems enhance soil properties and promote hydrological function (Savory and Parsons 1980; Savory 1988). These hydrological responses to grazing are strongly contingent on community composition, with communities that provide greater cover and obstruction to overland flow, such as midgrass-dominated communities having greater hydrological function, including infiltration rate, than shortgrass-dominated communities (Wood and Blackburn 1981b; Thurow 1991).

Grazing System. Short-duration rotational grazing systems decreased soil hydrologic function at heavy to very heavy stocking rates, compared to continuous and deferred-rotation grazing systems at moderate to light stocking rates. The negative changes in vegetation and soil properties controlling infiltration, runoff, and soil loss due to heavy stocking rates generally cannot be overcome by grazing system. These collective results strongly refute claims that animal trampling associated with high stocking rates or intensities under intensive rotational grazing systems enhance hydrological function (Savory and Parsons 1980; Savory 1988).

There is evidence that soil hydrological functions degraded by heavy stocking rates can recover with prolonged rest (i.e., ≥ 1 yr). Thus, rotational grazing may maintain higher soil hydrologic function than continuous grazing

at heavy to very heavy stocking rates if the deferment period is sufficient (i.e., ≥ 1 yr). Similarly, moderately stocked continuous or rotational grazing may maintain a consistently higher level of hydrologic function compared to periodic heavy stocking followed by prolonged deferment for hydrologic recovery.

A few studies have directly examined grazing systems (deferred rotation, rest rotation, and rotational deferment) in comparison with continuous grazing. At moderate stocking rates, at which most extensive rotational systems were studied, rotational grazing systems lead to similar or improved soil hydrologic function compared to moderate continuous grazing (Ratliff et al. 1972; McGinty et al. 1979; Wood and Blackburn 1981b, 1984). As evidenced by Wood and Blackburn (1981) and Thurow et al. (1986), these hydrological responses to grazing system appear to be strongly contingent on plant community composition, with midgrass-dominated communities having greater hydrological function than shortgrass-dominated communities. Gifford and Hawkins (1976) emphasize the importance that range condition or plant community composition has on the hydrological function of a site through time in response to grazing system.

Improve or Maintain Riparian and Watershed Function

There is clear consensus that livestock grazing can degrade riparian plant communities, hydrologic function, and associated ecosystem services. Considerable management attention has been directed toward prescribed grazing practices with the intent to restore, enhance, or maintain rangeland riparian areas. As with upland habitats, it is clear that grazing intensity is a major factor determining riparian response to grazing management. Increased grazing intensity is generally associated with detrimental effects on riparian plant community composition and productivity as well as physical degradation of riparian soils and stream channels. These primary effects can lead to secondary negative effects on stream hydrologic functions, which can cascade to loss of services, such as fish habitat, flood attenuation, and provisioning of clean water. Management of grazing intensity is a viable conservation practice for riparian areas. Season of grazing also determines livestock grazing effects on riparian plant communities, particularly woody plants, and can be managed to conserve riparian habitats and their associated services. Livestock distribution practices such as water developments, supplement placement, and herding are effective means of managing the intensity and season of livestock grazing in riparian areas. Livestock exclusion is an effective practice to stimulate immediate recovery for riparian plant communities degraded by heavy grazing. While the individual effects of some prescribed grazing components (e.g., timing, intensity, and rest) on riparian habitats have been examined, few studies have rigorously examined the effects of different grazing systems on riparian habitats. The effectiveness of grazing management practices on the conservation of riparian habitats is covered in depth in the chapter on riparian herbaceous cover (George et al., this volume).

Reduce Accelerated Soil Erosion and Maintain or Improve Soil Condition

Soil vegetative cover is widely recognized as a critical factor in maintaining soil surface hydrologic condition and reducing soil erosion (Gifford 1985). High stocking rates, regardless of grazing system, that reduce soil surface vegetative cover below a site-specific threshold will increase detachment and mobilization of soil particles due to raindrop impact, decrease soil organic matter and soil aggregate stability, increase soil surface crusting and reduce soil surface porosity, and thus decrease infiltration and increase soil erosion and sediment transport (Blackburn 1984). Regardless of grazing system, sufficient vegetative cover, critical soil cover, or residual biomass must remain during and following grazing to protect soil surface condition (e.g., porosity, aggregate stability, and organic matter content) and dependent hydrologic properties (e.g., infiltration). Site-specific vegetation cover requirements will vary depending on cover type (e.g., vegetation, litter, or rock), soil type, rainfall intensities, and water quality goals (Gifford 1985).

The majority of research examining soil surface hydrologic response to grazing has focused on infiltration or proxies for infiltration, such

“Increased grazing intensity is generally associated with detrimental effects on riparian plant community composition and productivity as well as physical degradation of riparian soils and stream channels.”

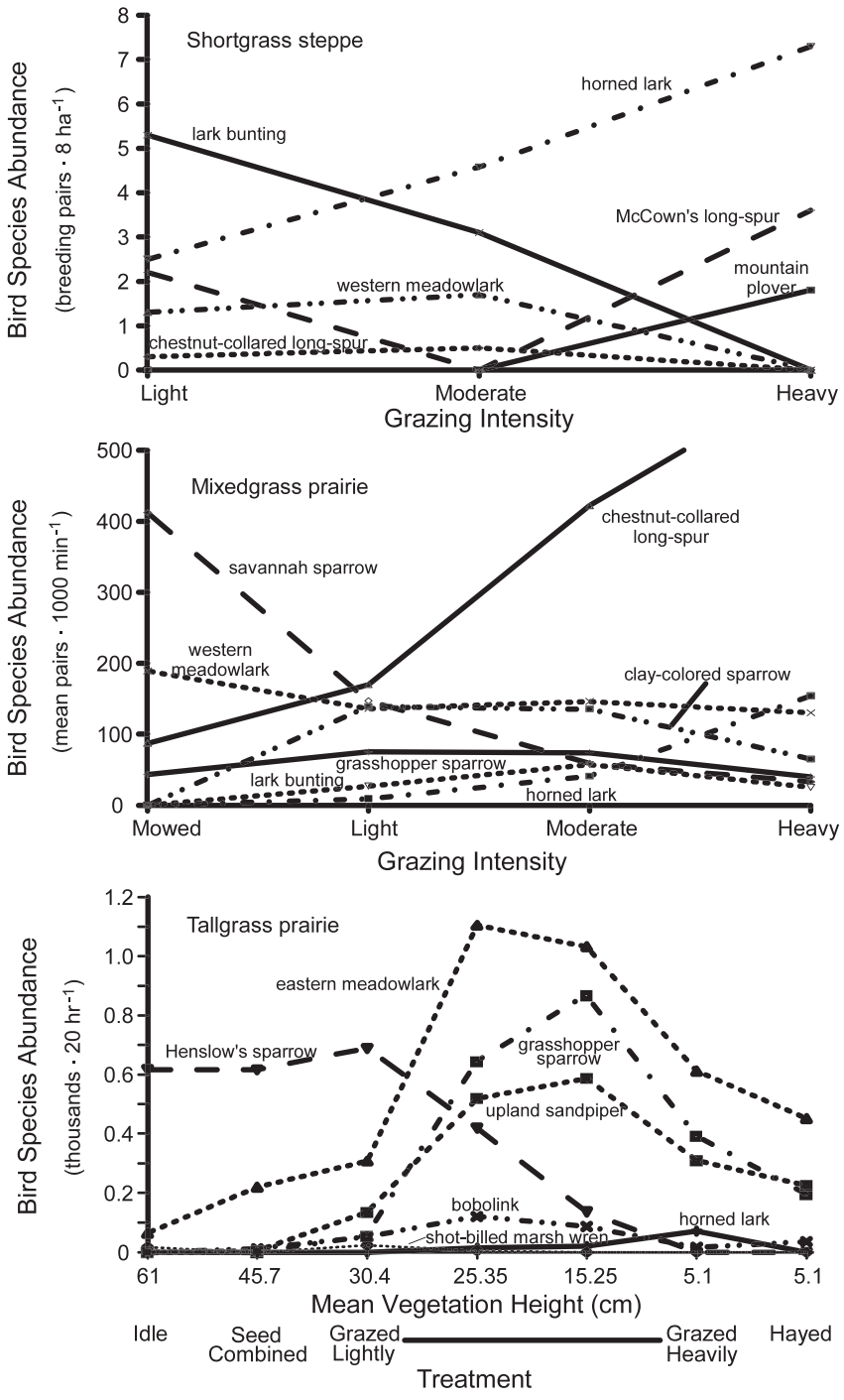


FIGURE 4. Bird species responses to grazing intensity/management treatments for (**A**) shortgrass steppe, (**B**) mixed-grass prairie, and (**C**) tallgrass prairie. Drawn from data in Giezentanner (1970), Skinner (1975), Kantrud (1981), Kantrud and Kologiski (1982), and Milchunas et al. (1998).

as dry bulk density and soil penetrability. A handful of studies have examined soil loss. Increased stocking rates from nongrazed to very heavy are associated with increased soil loss. As with infiltration results, light and moderate stocking rates are generally not different. There is no consistent result for the effect of grazing system on soil loss; in some cases, continuous systems are reported to have less soil loss, and in other studies, rotational systems are reported to have less soil loss. Most of these studies are confounded by comparisons of different stocking rates among systems, and several report that grazing system effect depended on plant community (e.g., shrub understory vs. interspace). There is no compelling evidence that rotational grazing strategies can reduce soil loss. Soil vegetative cover (responding to stocking rate) and inherent soil characteristics are key variables determining site scale soil loss (Pierson et al. 2002).

Improve or Maintain the Quantity and Quality of Food and/or Cover Available for Wildlife

Stocking Rate. Livestock and wildlife may directly compete for plant food resources, and livestock grazing can alter the composition, productivity, and quality of plant food resources. Grazing can alter community structure through removal of recent production and through longer-term effects on plant community composition and productivity. Cover represents an important component of wildlife habitat for escape and concealment from predation as well as for thermal regulation. Cover requirements for specific wildlife species often vary within a season and stage of life cycle (e.g., nesting vs. foraging). Bird (MacArthur 1965; Wiens 1969; Cody 1985), rodent (French et al. 1976; Grant and Birney 1979; Geier and Best 1980; Grant et al. 1982; Kerley and Whitford 2000), lagomorph (Flinders and Hansen 1975), and lizard (Pianka 1966) community composition and diversity are often closely correlated with vegetation structure. Direct behavioral interactions between livestock and wildlife are another potential means by which grazing may affect wildlife populations. Social avoidance can preclude the use of otherwise suitable habitat, and it can be influenced by the numbers of livestock present (Roberts and Becker 1982; Stewart et al. 2002). Trampling of nests represents another possible mechanism of negative interaction between livestock and ground-nesting birds that increases with stocking rate (Jensen et al. 1990b).

There are fewer studies documenting the responses of specific wildlife species or groups to stocking rate or grazing intensity than there are for plant communities. Therefore, studies published in the gray literature, including symposia and technical reports, have been included, but theses, dissertations, or non–data-based publications have not. Limited data availability also requires that inferences be drawn from individual studies rather than groups of studies, as has been done in other sections of this chapter. Wildlife responses are grouped into reptiles, birds, small mammals, and large ungulates to more effectively assess their potentially unique responses and interactions with livestock grazing.

Reptiles. Ten studies reported on lizard communities in grazed versus ungrazed treatments, but only one study assessed lizard populations over five grazing intensities in Arizona (Jones 1979, 1981). The largest negative effect of heavy grazing on lizard density was found in Sonoran Desert grassland (–63%), followed by mixed scrub–dry wash (–54%), chaparral (–41%), and cottonwood–willow riparian (–20%), with no difference in desert scrub. Greater species richness was observed in lightly compared to heavily grazed desert grassland and cottonwood–willow riparian habitat, with no difference in the other three communities. The effects of grazing on lizard communities were related to differences in the cover of short (< 0.3 m) vegetation structure and litter cover, but not necessarily total vegetation cover. While lizard responses to grazing may be expected to be more pronounced than for other groups of organisms because of their relatively specific microhabitat requirements, there are insufficient studies over grazing intensities for generalizations to be drawn.

Birds. Bird responses to stocking rate are well recognized as being species dependent and can be positive, negative, or neutral within any one location and treatment comparison (Bock et al. 1993; Saab et al. 1995; Knopf 1996). Unfortunately, most passerine bird studies have compared only grazed and ungrazed communities, and the intensity of grazing is often not reported. Derner et al. (unpublished data) reviewed 27 bird studies/habitats from the literature, and only 10 included more than one grazing intensity in addition to the long-term ungrazed community. The abundance of individual species within a site can be strongly affected by grazing intensity. For example,

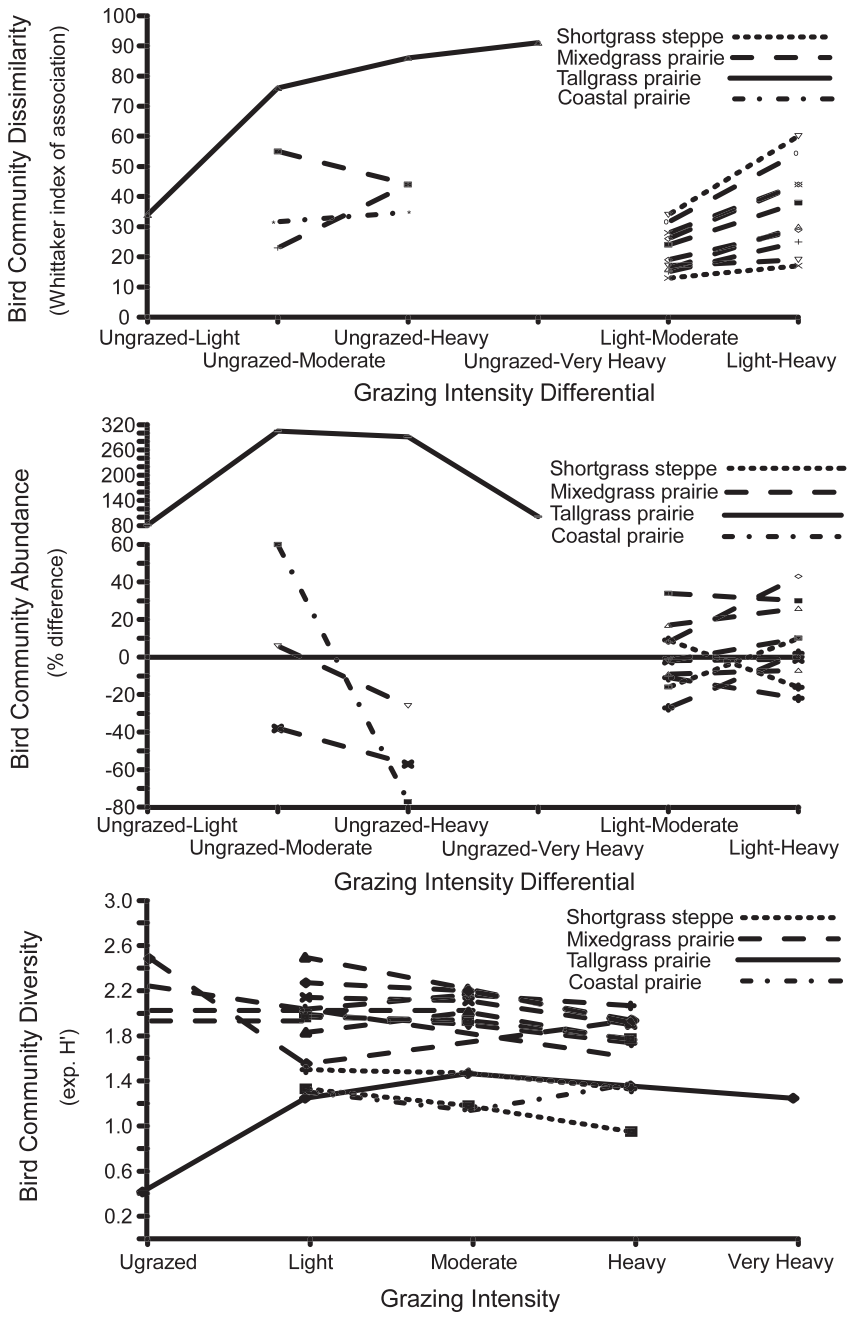


FIGURE 5. Bird community (**A**) dissimilarity (Whittaker [1952] index of community association), (**B**) abundance (% difference between grazing intensity differential), and (**C**) diversity (H′) across grazing intensity gradients for North America studies. Dissimilarity index values range from 0.0 to 1.0, with a value of 0 indicating both treatments having all species in common and in the same proportions (0% dissimilar) and a value of 1.0 indicating no species in common (100% dissimilar). Data from Giezentanner (1970), Johnson and Springer (1972), Skinner (1975), Grzybowski (1980, 1982), Kantrud (1981), Kantrud and Kologiski (1982), and Milchunas et al. (1998).

TABLE 1. Bird community dissimilarity, abundance (numbers), diversity, richness, and dominance in response to grazing averaged by region, evolutionary history of grazing, plant community life form, and plant community type. Forests were not included in region or evolutionary history categories. Plant community types are for major groupings or those with more than one comparison (from Derner et al., unpublished data).

Birds	Precipitation (average mm yr^{-1})	Dissimilarity (index)	Abundance (high grazed % low grazed)	Diversity (high grazed /low grazed)	Richness (high grazed/low grazed)	Dominance (high grazed/low grazed)	N
By region							
Great Plains	487	0.40	38	1.18	1.02	1.12	38
Southwest[1]	362	0.54	3	0.91	0.90	1.29	4
Northwest[1]	154	0.54	–22	1.25	1.14	0.83	6
Other grasslands	—[2]	0.54	–33	1.10	0.93	0.96	10
By evolutionary history							
Short history	617	0.53	–27	1.06	0.90	0.98	21
Long history	487	0.40	38	1.18	1.02	1.13	37
By life form							
Grassland	483	0.43	30	1.18	1.02	1.09	40
Shrubland	291	0.45	–5	1.10	1.03	1.01	11
Forest	1 182	0.52	–45	0.96	0.68	1.06	7
By community type							
Shortgrass steppe	357	0.29	36	0.94	0.83	1.13	6
Mixed-grass prairie	416	0.35	–2	0.91	0.90	1.25	23
Tallgrass prairie	988	0.61	217	2.52	1.71	0.59	6
Fescue grassland	383	0.60	–31	1.06	1.00	1.09	2
Coastal prairie	—[2]	0.42	–9	0.96	0.54	0.72	2
Southwest grassland	362	0.54	3	0.91	0.90	1.29	4
Shadscale shrubland	154	0.42	–31	1.49	1.43	0.62	2

[1]Northwest includes the Great Basin and all communities west of the Rocky Mountains, except for Arizona, New Mexico, and southern California, which are considered Southwest.

[2]Number of sites reporting precipitation too few to provide a reasonable mean.

horned larks respond positively to increasing grazing intensity in shortgrass steppe, while lark buntings respond negatively (Fig. 4A). Chestnut-collared long-spurs respond positively to increasing grazing intensity in mixed-grass prairie, while savannah sparrows respond negatively (Fig. 4B). The greatest abundance of bird species in tallgrass prairie occurred at intermediate intensities of grazing (Fig. 4C). While species within a site respond differently to grazing intensity, a particular species may also have a varied response among sites. Knopf (1996) suggested that birds may not be generally classified as increasers or decreasers in response to grazing, but that individual species responses to grazing may vary over gradients of potential vegetation structure or aboveground primary production. Although there are examples for regional differences in bird species response to grazing, Derner et al. (unpublished data) concluded that data over gradients of grazing intensity and regional gradients of primary production are too limited to produce good models of bird preferences for particular grazing intensities at particular levels of primary production. Reviews by Bock et al. (1993) and Saab et al. (1995) provide tables of bird species by region within the western United States that show general positive, negative, primary productivity–dependent, or neutral/mixed/uncertain responses to grazing.

At the community level, the change in bird community composition relative to the ungrazed or lightly grazed condition usually increased with increasing grazing intensity (Fig. 5A; Table 1). However, dissimilarity was generally greater when the communities were ungrazed compared to lightly or moderately grazed than when grazing intensity further increased to moderate or heavy. Total bird community abundance showed both positive and negative responses with increasing grazing intensity across and within community types as anticipated (Fig. 5B). Bird community diversity was generally slightly negative with increasing grazing intensity (Fig. 5C). Exceptions were observed for one tallgrass prairie study and some mixed-grass prairie sites where slightly greater diversity occurred at intermediate levels of grazing intensity. In addition to these general diversity patterns, management decisions need to explicitly evaluate the specific habitat needs of bird species of concern.

Most studies of grazing effects on upland game birds (gallinaceous birds) addressed ungrazed versus grazed conditions rather than grazing intensity gradients, much like research for all other wildlife groups. Based on two studies, wild turkeys prefer ungrazed/lightly grazed vegetation and avoid moderately/heavily grazed areas. Similarly, heavy grazing was consistently detrimental to sharp-tailed grouse (three subspecies) because of a loss of nesting cover and tree and shrub density (based on 10 studies reviewed in Kessler and Bosch 1982). There are contrasting positive and negative results from ungrazed/grazed studies for sage grouse and prairie chickens, but sage grouse appear to prefer light/moderate grazed areas over heavy grazed areas, but very high cover in some ungrazed habitat may be avoided as well (some reviewed in Beck and Mitchell 2000). Historical evidence suggests that grazing is detrimental to quail species in the southwestern United States, but recent studies indicate that light to moderate grazing intensities may be beneficial to Mearn's quail by increasing availability of food resources. Montezuma quail prefer high grass cover and tree density, while scaled quail prefer high grass cover and low tree density. In contrast, five studies of bobwhite quail in Texas (see Bryant et al. 1982) suggest that grazing is beneficial if intensities are not too high. In summary, heavy grazing most often results in loss of cover below some optimal level for gallinaceous birds, although light grazing may be beneficial under some circumstances.

Small Mammals. Small mammals can be sensitive to changes in vegetation structure, but they may also be affected by grazing

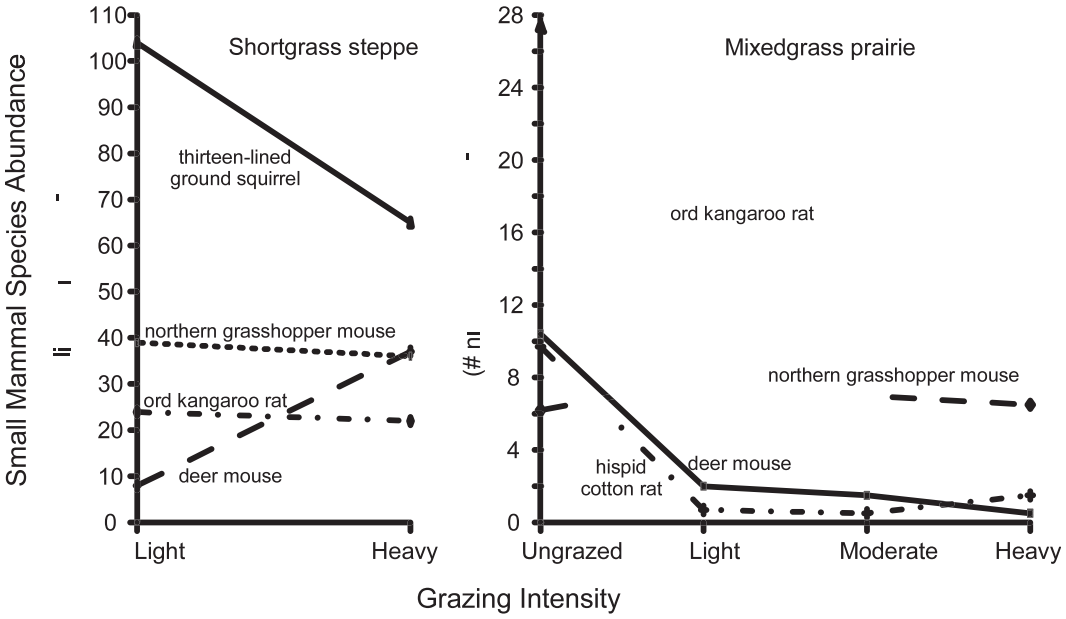


FIGURE 6. Rodent species abundance across grazing intensities in (**A**) shortgrass steppe and (**B**) mixed-grass prairie. Drawn from data in McCulloch (1959) and Grant et al. (1982).

TABLE 2. Rodent community dissimilarity, abundance (numbers), diversity, richness, and dominance in response to grazing averaged by region, evolutionary history of grazing, plant community life form, and plant community type. Forests were not included in region or evolutionary history categories. Plant community types are for major groupings or those with more than one comparison (from Derner et al., unpublished data).

Rodents	Dissimilarity (index)	Abundance (high grazed % low grazed)	Diversity (high grazed/low grazed)	Richness (high grazed/low grazed)	Dominance (high grazed/ low grazed)	Unique species (high grazed/low grazed)	N
By region							
Great Plains	0.35	–27	0.99	0.89	1.11	–2.0	14
Southwest[1]	0.34	24	0.89	0.85	1.41	–1.0	6
Northwest[1]	0.43	8	0.81	0.95	1.60	–0.8	19
By evolutionary history							
Short history	0.41	12	0.83	0.93	1.55	–0.9	25
Long history	0.35	–27	0.99	0.87	1.11	–2.0	14
By life form							
Desert	0.18	–43	0.85	0.73	1.40	–1.5	2
Grassland	0.34	13	0.73	0.92	1.30	–1.4	4
Shrubland	0.38	–0	0.73	0.76	1.62	–1.7	12
Savanna	0.43	14	0.92	1.51	1.53	1.0	3
Forest	0.30	–55	0.82	0.75	0.96	–1.0	2
By community							
Shortgrass steppe	0.19	–9	1.24	1.0	0.68	0.0	1
Mixed-grass prairie	0.32	–18	0.81	0.78	1.29	–2.9	9
Grassland	0.47	–13	0.79	0.62	1.53	–5.8	4
Sand sage shrub	0.19	–23	0.82	0.91	1.10	–0.6	5
Tallgrass prairie	0.48	–50	1.34	1.1	0.80	–0.5	4
Desert grassland	0.41	58	0.91	0.92	1.41	–0.8	4
Shadscale shrubland	0.53	1	0.57	0.6	2.36	–2.0	2
Atriplex shrubland	0.52	–27	0.63	0.96	1.81	0.0	2
Sagebrush shrubland	0.30	30	0.76	0.86	1.49	–1.5	6
Northwest grassland[2]	0.34	13	0.73	0.95	1.30	–0.3	4

[1]Northwest includes the Great Basin and all communities west of the Rocky Mountains, except for Arizona, New Mexico, and southern California, which are considered Southwest.

[2]See Savanna for Northwest savannas.

induced modification of seed and arthropod food resources. Derner et al. (unpublished data) reviewed 24 rodent studies/habitats from the literature, and only six included more than one grazing intensity in addition to long-term ungrazed exclosures. The responses of individual species of small mammals to grazing intensities are similar to birds, but they differ from birds at the community level. Like birds, some rodent species are favored by grazing, some decline, and others are relatively neutral (Fig. 6 A and B). The response of some

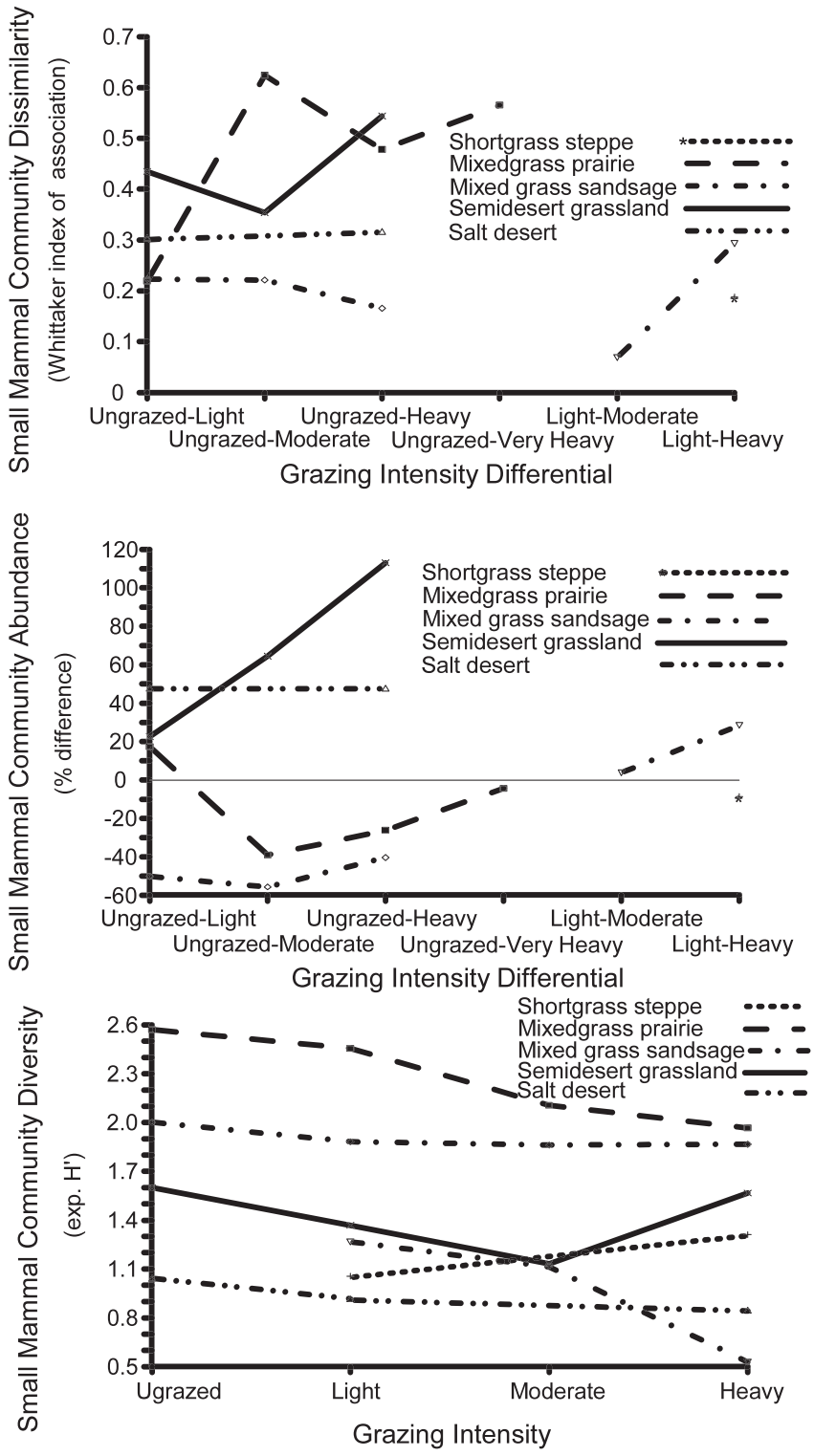


FIGURE 7. Rodent community (**A**) dissimilarity (Whittaker [1952] index of community association), (**B**) abundance (% difference between grazing intensity differential), and (**C**) diversity (H') across grazing intensity gradients for North America studies. Dissimilarity index values range from 0.0 to 1.0, with a value of 0 indicating both treatments having all species in common and in the same proportions (0% dissimilar) and a value of 1.0 indicating no species in common (100% dissimilar). Data from Frank (1940), Smith (1940), McCulloch (1959), Grant et al. (1982), Rice and Smith (1988), and Bich et al. (1995).

species to grazing intensity can be substantial. Generalizations concerning rodent responses to livestock grazing intensity are less developed than those for birds, in part because of fewer studies but also because of less consistent population responses.

Derner et al. (unpublished data) assessed the number of rodent species unique to various grazing intensities to evaluate the general patterns of declining rodent diversity with increasing grazing intensity. Greater numbers of species were likely to be captured on ungrazed or lightly grazed communities than on moderately or heavily grazed communities. However, 19 of 41 cases also displayed species unique to the more intensively grazed communities as well, but in only five cases was the total number of unique species greater on the more intensively grazed community. The net effects by region and evolutionary history were unique and unexpected. The numbers of unique species associated with heavy grazing were smaller in the Great Plains than in the Southwest or Northwest and in ecoregions with long rather than short evolutionary histories of grazing (Table 2). Deserts, grasslands, and shrublands displayed somewhat similar reductions in rodent species with increasing grazing intensity, and the losses were greater compared to savannas and forests. The greatest reductions in rodent species with increasing grazing intensities occurred in mixed-grass prairie. In general, no consistent trends could be discerned for changes in rodent species composition with grazing intensity relative to ungrazed or lightly grazed condition (dissimilarity; Fig. 7A) or abundance; Fig. 7B). Rodent diversity generally declines or is unchanged with increasing grazing intensity, with the exception of a shortgrass steppe study (Fig. 7C). Declines in rodent diversity with grazing intensity were only small to moderate when they were observed.

Bird responses to grazing are highly species specific and positive, negative and neutral outcomes occur. (Photo: USDA: Gary Kramer)

Other Small Mammals. Heavily grazed or "overgrazed" communities are generally preferred over ungrazed or lightly grazed communities by black-tailed jackrabbits in eastern Texas (Taylor and Lay 1944), the Mojave Desert in California (Brooks 1999), the sand hills of Colorado (Sanderson 1959), and southern Arizona (Taylor et al. 1935) and by Great Plains jackrabbits in mixed-grass (Smith 1940) and tallgrass prairie (Phillips 1936) of Oklahoma. Schmutz et al. (1992) observed that rabbits became more abundant as range conditions deteriorated in desert grassland. MacMahon and Wagner (1985) suggested that many areas of the Chihuahuan and Sonoran deserts initially altered by fire suppression and livestock grazing do not return to previous conditions when large herbivores are excluded because lagomorphs and rodents, favored by the initial changes, maintain the vegetation at early seral stages. In contrast, Flinders and Hansen (1975) found that cottontail rabbits were more abundant in moderately than in either lightly or heavily grazed shortgrass steppe, white-tailed jackrabbits showed no preference, and black-tailed jackrabbits were more abundant in lightly and moderately compared to heavily grazed communities. Changes beneficial to rabbits with increasing grazing intensity include increased rabbit mobility and improved forage due to increases in annuals.

Wild Ungulates. There is a large body of research addressing dietary and habitat use overlap between livestock and deer and elk. In general, high dietary overlap is observed between cattle or sheep and elk, compared with much lower overlap between cattle or sheep and deer (Skovlin et al. 1968; Mackie 1970; MacCracken and Hansen 1981; Berg and Hudson 1982; Loft et al. 1991). However, dietary overlap between deer and cattle can increase with increasing intensity of cattle grazing (Mackie 1981; Vavra et al. 1982; Severson and Medina 1983). Habitat use is often separated in time because of seasonal migrations of deer and elk and in space because of topography or cover requirements (Skovlin et al. 1968; Mackie 1970; Berg and Hudson 1982). For example, mule deer, elk, and cattle observations on slopes steeper than 10 degrees averaged 50%, 42%, and 18%, respectively. Dietary overlap between domestic and native herbivores is generally greatest during the period in which the herbivores are most nutritionally stressed (Olsen and Hansen 1977; Mackie 1981), and habitat overlap is most likely to occur when wildlife are at lower elevations during winter, which often represents the period of greatest nutritional stress (Wallmo et al. 1981).

Increasing grazing intensities by livestock are likely to create a bottleneck in the quantity and quality of forage for wild ungulates during nutritionally stressed periods (e.g., winter or drought). More generalist, large-rumen livestock are better able to utilize dormant grass forage than deer under conditions of low forage availability in heavily compared to moderately or lightly stocked pastures (MacMahan and Ramsey 1965). Dietary overlap between cattle

and pronghorn is low, and Schwartz et al. (1977) found that pronghorn were able to maintain seasonal diet qualities on long-term pastures heavily grazed by cattle similar to what they did on lightly grazed pastures in shortgrass steppe under nondrought conditions. In contrast, a pronghorn die-off was attributed to very heavy grazing by domestic animals during a drought (Hailey et al. 1966). Other studies of livestock grazing effects on pronghorn populations also show mixed responses.

In contrast, habitat overlap is a prerequisite to facilitation of one herbivore by another. Positive or facilitative effects of livestock grazing on associated wildlife species may result from a reduction in the amount of unpalatable, standing dead material (Short and Knight 2003) or increased protein content and digestibility of forage available late in the season (Clark et al. 2000; see below). Both competition and facilitation can act simultaneously, and competition can be the strongest factor (Hobbs et al. 1995). Longer-term facilitative relationships may be based on a dichotomy in diet preference of grass versus forbs and shrubs. For example, grazing by deer and livestock can potentially shift community composition toward a composition favored by other species of herbivores.

Grazing can potentially be used as a tool to enhance wildlife populations, and this may be particularly true when season of grazing or deferment of grazing is used to meet specific wildlife goals. In some situations, wildlife and livestock may overlap in habitat use only during particular times of the year. For examples, breeding birds may nest only during spring/early summer and require specific conditions during that time. Elk and deer may move down from forested mountainous habitat during the winter to occupy foothills and plains more likely to be used for livestock grazing. Grazing may be imposed or deferred, depending on cover and foraging requirements of specific species. Some waterfowl or some upland game species require dense nesting cover, whereas some birds, such as mountain plover or curlews, choose nesting sites with very little cover and will not nest in ungrazed or lightly grazed habitat. Many of the examples of season-of-use studies come from wildlife refuges or experimental sites where livestock use is optional and management options are flexible.

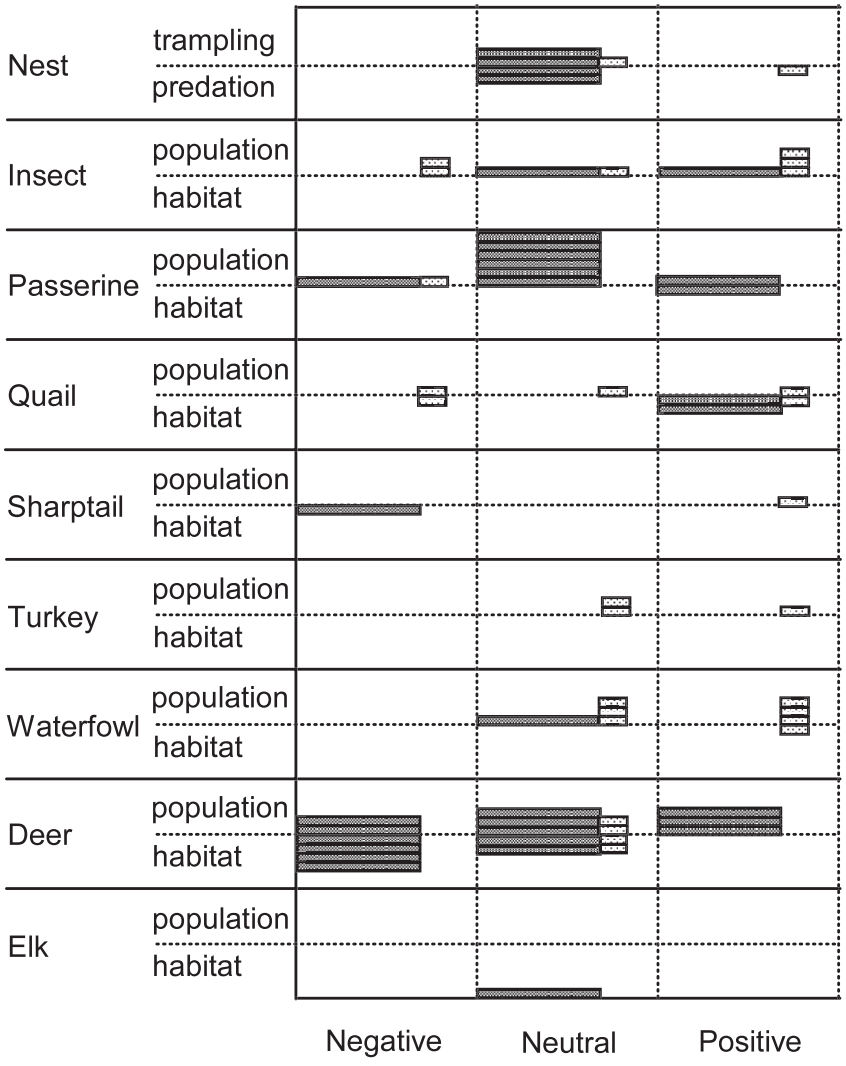


FIGURE 8. Responses of nine wildlife categories to rotational compared to continuous livestock grazing systems summarized as neutral, negative, or positive. Each wildlife category is subdivided into a population (upper) and a habitat (lower) section to indicate the mechanism of livestock impact. The large dark bars represent studies without confounding experimental designs, and the small lightly shaded bars represent studies with design problems. Each study may have one to four response variables, so each bar does not represent a single study.

Successful use of season of grazing may result from a facilitation effect of grazing by livestock on forage for other ungulates. Alpe et al. (1999) showed that early summer grazing improved forage quality for wild ungulates in autumn and winter if livestock was removed in time to allow sufficient regrowth, but that late season livestock grazing decreased forage quality for wild

ungulates. Successful use of season/deferment of grazing may also be possible when pastures that are not frequently used by wildlife are available; otherwise the removal of livestock from one pasture must outweigh the effects of increased stocking rate in adjacent pastures. Even in these cases, experimental outcomes can be neutral, positive, negative, or mixed, depending on wildlife species or timing of grazing (Medin 1986; Alpe et al. 1999; Mathis et al. 2006).

Grazing Systems. Wildlife responses to rotational and continuous grazing at relatively similar grazing intensities and within similar plant communities are evaluated in this section. Studies investigating different pastures within the same grazing system are also considered separately and clearly identified when used. Studies are organized by wildlife taxonomic groups and summarized across all groups and mechanisms for positive, neutral, or negative responses to grazing system.

Birds. Although passerine birds represent the most studied group of wildlife in response to grazing intensity (see section above), only two published studies of rotational compared to continuous grazing were located. However, there are a number of unpublished theses, studies within pastures of an individual system or that compare rotational grazing with ungrazed communities that were not considered here. The dissertation of Kempema (2007) was unique because it assessed several grazing periods of increasing duration, so it is summarized here. These studies report that passerine responses to grazing systems compared to continuous grazing were most often neutral (Fig. 8). The rotational systems had the least vegetative heterogeneity at both small and large spatial scales because of the reduced capacity for selective grazing at the bite and patch scale by livestock compared to continuous grazing. This was accompanied by a decrease in bird species richness with decreasing duration of grazing (long-continuous richest). In contrast, the short-duration system had the highest densities of the most species. For most bird species (11), there was no significant grazing system effect on density, and for the three species that showed significant density effects, the responses were both positive and negative. Nest success was also similar among the three grazing systems. The small number of passerine studies specifically conducted in grazing systems precludes the development of general conclusions.

Ungulate responses to grazing are equivocal so that no broad conclusions can be drawn. (Photo: USDA: Gary Kramer)

Grazing systems studies on gallinaceous birds frequently evaluated nest trampling or nest predation, often conducted with artificial nests. All studies without confounded designs show neutral responses to grazing system (Fig. 8). Larger densities of livestock in smaller pastures of rotational systems do not appear to increase trampling losses under the densities and in the habitats studied. A higher density of livestock in some pastures and longer rests in others appears to produce a similar mean effect. Trampling of nests is often found to increase linearly with stocking density (Bientema and Müskens 1987; Paine et al. 1996; Kempema 2007), and the ecological significance of nest trampling is greater in more productive ecosystems that support greater stocking density. In contrast, Koerth et al. (1983) found trampling losses of nests to be similar between short-duration and continuous grazing, even though the short-duration system was stocked at a higher rate (5.3 vs. 8.0 ha · steer^{-1}). Nest trampling may not be linear with stocking density because livestock

may travel less in smaller pastures (Koerth et al. 1983 and citations therein). Alternatively, a reduction in diet selection may increase search time exploring new pastures during repeated rotations (Spalinger and Hobbs 1992; Wilmshurst et al. 1999), and large herd size may result in more temporally constant activity levels among livestock (Paine et al. 1996).

Direct studies of population or habitat responses between grazing systems are few, but positive responses have been reported for rotational grazing systems compared to continuous grazing for bobwhite quail in response to increased bare ground and greater forb densities (Fig. 8). There are too few studies for sharptail or turkey to draw any meaningful conclusions concerning the effect of grazing system. No grazing systems studies were found for prairie chickens or sage grouse, but some management recommendations have been made, including multiyear periods of rest to restore vegetative cover (Hagen et al. 2004).

Vegetative cover is an important habitat requirement for waterfowl, although very dense vegetation can be detrimental to nest-site selection (Kantrud 1990). Ignatiuk and Duncan (2001) observed no difference in duck nest success in an extensive study of once-over rest-rotation or deferred-rotation systems and continuous grazing, while additional studies compared only pastures within grazing systems or conditions following changes in grazing regime (Fig. 8). When rest periods were from 1 to 3 yr, Gilbert et al. (1996) observed increasing duck nest densities with increasing years of rest, and regression analyses suggested that a 6- to 7-yr rest would be necessary for recovery to that of an ungrazed condition. Other waterfowl studies with confounding experimental designs also suggest that long rest periods may be beneficial, but there are too few waterfowl studies of grazing systems to form robust conclusions.

Large Ungulates. Grazing systems research has been conducted with elk, deer, and pronghorn antelope, but the pronghorn study compared rest-rotation only with ungrazed pastures. Eight studies that included 18 response variables were found comparing grazing systems with continuous grazing for deer. The most common deer response to grazing system was negative, followed by neutral and then positive responses (Fig. 8). However, population-level responses for deer were equally split between positive and neutral for rotational compared to continuous grazing. Studies assessing habitat characteristics important to deer were most often negative in rotational grazing systems compared to continuous. Only one grazing system study reported on social avoidance by deer, and it showed deer–livestock competition in the short-duration system compared to continuous grazing that was attributed to habitat modification rather than deer leaving the pasture (Cohen et al. 1989). Responses of deer to rotational systems are generally mixed so that no clear trends can be established.

Only one study was located that directly compared elk responses in a deferred-rotation grazing system compared to season-long grazing and found no significant response when averaged over grazing intensities (Fig. 8) but did find a highly significant interaction of grazing system with grazing intensity (Skovlin et al. 1968, 1975, 1983). Elk preferred season-long to deferred rotation at the light grazing intensity, but preferred deferred to season-long rotation at the high grazing intensity. Elk utilized individual plants that had not been grazed by cattle, and cattle use of numbers of individual plants at the low grazing intensity was greater under the rotation system. Forage quantity and preference for areas receiving little or no prior current-year use by livestock can regulate elk movement across larger landscapes as well (Mackie 1970). These results and the observation that elk preference strongly increased with decreasing grazing intensity even from light to ungrazed treatments are in accordance with the within-system studies showing a high degree of elk sensitivity to livestock grazing intensity and selection for ungrazed units or treatments, unutilized/little utilized areas within grazed pastures, and ungrazed individual plants. However, the studies within various pastures of a single rotational grazing system are often cited to support rotational grazing as benefiting elk populations. Three of these studies found no social avoidance between elk and livestock for selection of ungrazed pastures. Livestock grazing facilitated use by elk the year following grazing in two studies, and elk avoided currently and previously grazed pastures in the other study.

> “…the most frequent wildlife response was no differences between continuous and rotational grazing systems, with the remaining cases equally divided among positive and negative.”

“The response of soil organic carbon to stocking rate is equivocal, based partially on the limited number of investigations that have been conducted.”

Summary of Wildlife and Grazing Systems. The limited number of available studies does not permit generalizations concerning wildlife responses to grazing systems and when or where or for which species positive, negative, or neutral responses may be predicted. There appear to be many false claims and few valid studies in the literature (Kirby et al. 1992), and this assessment applies to the literature addressing wildlife responses to grazing systems. Collectively, comparative wildlife responses to rotational and continuous grazing were that 17 showed no difference, eight were negative, and eight were positive (Fig. 8). These experimental data indicate that the most frequent wildlife response was no differences between continuous and rotational grazing systems, with the remaining cases equally divided among positive and negative. However, most wildlife groups showed mixed responses to grazing system, and it is clear that there are conditions where rotational grazing systems benefit a wildlife species or group, but the opposite response is documented as well. Much more is known about wildlife responses to grazing intensities than grazing systems, but even here the majority of studies assess grazed and long-term ungrazed communities, which are generally not relevant to prescribed grazing management (Krausman et al., this volume).

Manage Fine Fuel Loads to Achieve Desired Conditions

Grazing does reduce fine fuel loads, and it can therefore modify both fire frequency and intensity (Belsky and Blumenthal 1997; Briggs et al. 2002; Fuhlendorf and Engle 2004). This interpretation is supported by the well-documented inverse relationship between stocking rate and aboveground herbaceous standing crop (Bement 1969; Milchunas and Lauenroth 1993; Manley et al. 1997; Derner and Hart 2007). It is often hypothesized that woody plant encroachment is partially a consequence of reduced fire regimes associated with livestock grazing (Scholes and Archer 1997; Swetnam and Betancourt 1998; Briggs et al. 2005). However, beyond these broad generalizations, there are only limited experiential data to support grazing as a means of fuel management (Belsky and Blumenthal 1997; Davies et al. 2010). This is perhaps not that surprising given that fire–grazing interactions are strongly influenced by site, year, season, and specific fire conditions (Davies et al. 2009).

Patterns of fire and grazing appear to be critically linked on the landscape (Fuhlendorf and Engle 2004). Grazing may increase the variability on fire occurrence by reducing the amount and increasing the heterogeneity of fine fuel distribution (Holdo et al. 2009). Grazed patches have less fine fuel that is less likely to burn than ungrazed patches that contain larger amounts of combustible fine fuel (Collins and Smith 2006; Kirby et al. 2007). However, grazing increased fuel homogeneity in a bunchgrass-dominated rangeland by reducing biomass of individual plants to a greater extent than biomass in the plant interspaces (Davies et al. 2010).

Weather and fuel conditions further increase the complexity of the relationship between fuel load and fire frequency and intensity. For example, fine fuel load is strongly correlated with fire intensity when fuel moisture is held constant, but when fuel moisture is low, intense fires can be carried by much lower fuel loads (Twidwell et al. 2009). This will be influenced by the season, time of day, and specific weather conditions associated with individual fires. It is no coincidence that most wildfires occur during extreme fire conditions; during these extreme conditions, fire can be carried by a wide range of fuel loads. Therefore, it should not be assumed that fire frequency and intensity decrease linearly with decreasing fuel loads resulting from greater grazing intensities.

The relative proportions of fine and coarse fuel loads can also influence the relationship between grazing and fire frequency and intensity. Woody plant encroachment is often associated with a reduction in the amount of fine fuel, but coarse fuel loads often increase substantially (Hibbard et al. 2001; Norris et al. 2001; Briggs et al. 2002). Although coarse fuels have higher ignition temperatures, closed-canopy woodlands can be highly flammable during extreme fire conditions. Therefore, the role of grazing as a tool for fuel management is generally supported, but it should be cautiously evaluated on a case-by-case basis because fire potential in influenced by interactions among several ecosystem variables (Fuhlendorf et al., this volume).

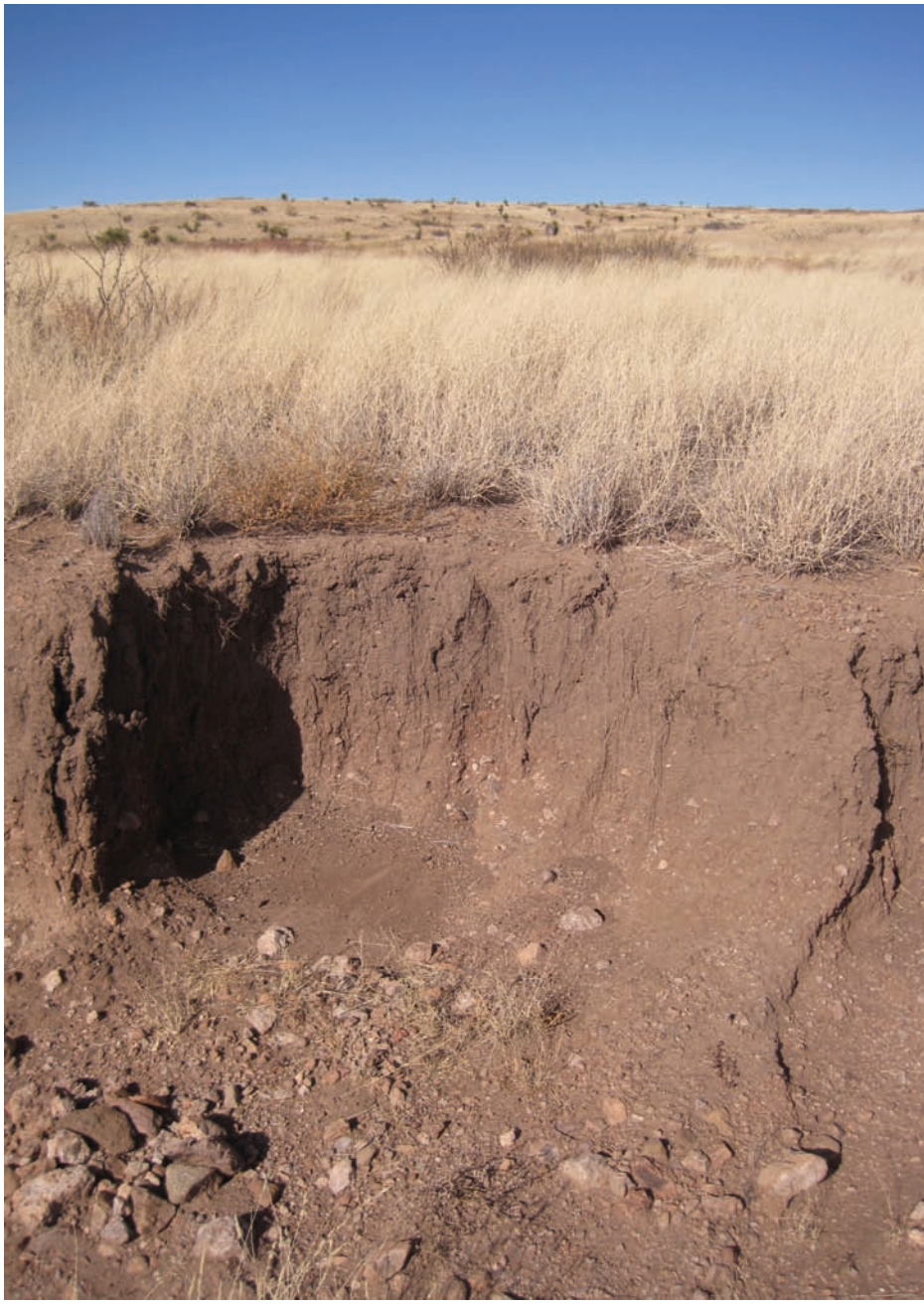
ASSOCIATED CONSIDERATIONS

Livestock Distribution

Animal selectivity and foraging behavior within landscapes has received considerable attention on rangelands (Bailey et al. 1996; Launchbaugh and Howery 2005). Herbivores naturally select preferred plants and landscape positions over others (Van Soest 1994), resulting in differential patterns of species use within communities and management units when stocking rates are not excessive and pastures are of sufficient size (Bailey et al. 1996; Launchbaugh and Howery 2005). Rangelands have traditionally been managed to increase uniformity of vegetation use by livestock and maximize livestock gains within the limits of individual animal performance and long-term ecosystem sustainability (Bement 1969). This management approach has been effective and sustainable from the standpoint of livestock and forage production (e.g., Hart and Ashby 1998), but it often does not mimic the pattern of historic disturbance regimes (Fuhlendorf and Engle 2001) or create habitat structure required for many grassland bird species (Knopf 1996; see Deferment and Rest section below). Livestock distribution and grazing behavior can be modified by adjusting the location of supplemental feed and water, implementation of patch burns, and herding (Williams 1954; Ganskopp 2001; Fuhlendorf and Engle 2004; Bailey 2005) in addition to the traditional practice of fencing.

Experimental data evaluating the most critical variables associated with livestock distribution were evaluated from 51 studies and two reviews. Treatment responses were categorized into 1) general distribution effects, 2) steep-slope use, 3) high-elevation use, 4) distance from water, 5) plant preferences, 6) uniformity of grazing, and 7) riparian use. All 51 studies were short term (< 5 yr), and the vast majority of them used cattle as the livestock species (41). Pasture sizes used in these investigations were generally large (22 > 200 ha). Recent investigations have incorporated technological advances involving GPS devices (e.g., collars) to track individual animal movement to provide spatial- and temporal-explicit use patterns. Strategies for modifying patterns of livestock distribution have shifted from specific practices (e.g., fences, salt, and water placement) to the modification of animal behavior (e.g., attractants, genetic selection, breeds, and type of animal) over the past two decades. Livestock distribution in response to specific conservation practices have received relatively little attention with the exception of prescribed burning (see Fuhlendorf et al., this volume).

The experimental data verify that many of the common assumptions regarding livestock distribution and preferences for specific sites



Rangelands play an important role in the global carbon cycle because of the large reservoirs of organic and inorganic carbon they contain. (Photo: Brandon Bestelmeyer)

Prescribed grazing must balance the forage demand of animals with the physiological requirements of plants to be sustainable. (Photo: USDA: Lynn Betts)

and conditions are valid. Water distribution (11 of 15 studies), steep slopes, and high elevations (13 of 17 studies) unequivocally influenced livestock distribution. Livestock by and large prefer riparian to upland areas (e.g., Bowns 1971; Smith et al. 1992; Howery et al. 1996, 1998), burned to nonburned areas (Coppedge and Shaw 1998; Biondini et al. 1999), previously grazed compared to ungrazed areas (Ganskoppp and Bohnert 2006), and fertilized to nonfertilized areas (Samuel et al. 1980). Range riding and/or herding of animals also effectively modified livestock distribution (Skovlin 1957; Bailey et al. 2008). A clear exception to these generalizations is that salt location has only a minor influence on grazing distribution within a growing season (five of seven studies; Ganskopp 2001). Standard approaches to modifying livestock distribution are warranted, but it appears that they can only minimize animal selection and preferences rather than completely eliminate them (Jensen et al. 1990a).

Grazing and Soil Organic Carbon

Rangelands play an important role in the global C cycle because of 1) an extensive land area, 2) large reservoir of sequestered C that could be released back into the atmosphere with improper management, 3) potential for high rates of soil organic carbon (SOC) accumulation by restoration of degraded rangelands, and 4) a vast pool of soil inorganic C as carbonates in semiarid and arid rangeland soils that may allow sequestration or release of CO_2 (Schuman et al. 1999; Derner and Schuman 2007; Svejcar et al. 2008). SOC sequestration is influenced by climate (Derner et al. 2006), biome type (Conant et al. 2001), management (grazing, N inputs, restoration, and fire; Follett et al. 2001; Mortenson et al. 2004; Derner and Schuman 2007; Bremer and Ham 2010; Pineiro et al. 2010), and environmental conditions (drought and climate change; Jones and Donnelly 2004; Ingram et al. 2008; Svejcar et al. 2008). Rangelands are typically characterized by short periods of high C uptake (2–3 mo · yr^{-1}), long periods of C balance or small losses (Svejcar et al. 2008), and climate-driven interannual variability in net ecosystem exchange (Zhang et al. 2010). Three main drivers that will control the fate of C sequestration in rangelands are 1) long-term changes in production and quality of above- and belowground biomass; 2) long-term changes in the global environment, such as rising temperatures, altered precipitation patterns, and rising CO_2 concentrations, that affect plant community composition and forage quality; and 3) effects of short-term weather conditions (e.g., droughts) and interannual variability in climate on net C exchange (Ciais et al. 2005; Soussana and Lüschert 2007; Ingram et al. 2008; Svejcar et al. 2008; Zhang et al. 2010).

Application of appropriate management practices, such as proper stocking rates, adaptive management, and destocking during drought conditions on poorly managed rangelands (113 M ha), could result in sequestration of 11 Tg C · yr^{-1}, and continuation of sustainable management practices on the remaining rangelands would avoid losses of 43 Tg C · yr^{-1} (Schuman et al. 2001).

SOC sequestration rates decrease with longevity of the management practice (Derner and Schuman 2007), indicating that ecosystems reach a "steady state" and that changes in inputs would be required to sequester additional C (Conant et al. 2001, 2003; Swift 2001). The response of SOC to stocking rate is equivocal, based partially on the limited number of investigations that have been conducted. Sixty-two percent (five of eight) of the investigations showed no response of SOC to stocking rate (Smoliak et al. 1972; Wood and Blackburn 1984; Warren et al. 1986a; Biondini et al. 1998; Schuman et al. 1999) with one showing a decrease (Ingram et al. 2008) and two showing an increase in response to increasing stocking rate (Manley et al. 1995; Reeder and Schuman 2002). The two investigations showing an increase in SOC with increasing stocking rate occurred in the northern mixed-grass prairie during a relatively wet period (Manley et al. 1995; Reeder and Schuman 2002). It has been demonstrated that increasing SOC in these grasslands may partially result from increasing dominance of the shallow-rooted, grazing-resistant species blue grama (*Bouteloua gracilis*), which incorporates a larger amount of root mass in the upper soil profile than do midgrass species that it replaces (Derner et al. 2006). In a global analysis, Milchunas and Lauenroth (1993) found that in 19 of 34 comparisons, SOC was less in grazed than ungrazed communities, and results were similarly mixed for root biomass.

Contributions of Individual Plant Research to Grazing Management

Many of the assumptions on which grazing management is founded originated from defoliation experiments conducted with individual plants. Suppression of plant photosynthesis, root growth cessation, support of regrowth by carbohydrate reserves, and regulation of tillering by apical dominance represent several of the major assumptions (Briske and Richards 1995). The relevance of these individual plant-based assumptions to grazing management has recently been questioned in an assessment of plant and animal production responses to grazing systems (Briske et al. 2008). In several instances, these plant-based assumptions have shown little correspondence with the outcomes observed in grazing systems. Since the development of these plant-based assumptions in the mid-20th century, some have been substantiated, but others have been refuted from the vantage point of greater scientific understanding derived from more sophisticated experimental techniques. Several plant-based assumptions that have been validated and invalidated are summarized below. Unfortunately, these assumptions often prevail long after they have been refuted by substantial experimental evidence.

Valid Plant-Based Interpretations. Numerous plant-based interpretations were developed early in the profession to cope with widespread overgrazing and rangeland degradation that prevailed in the late 19th and early 20th centuries. These were often based on observation and general inference because knowledge of plant physiology was very limited during this period and did not substantially improve until the mid-20th century. Several of the more important plant-based interpretations that have been supported by current science are summarized below.

Leaf Removal and Subsequent Growth. Photosynthetic leaf area provides the energy source for plant growth and reductions in leaf area suppress both plant photosynthesis and growth (Sampson 1923). This interpretation has been well supported with additional insights addressing the various contributions of leaf canopy position and leaf age (Caldwell et al. 1981; Gold and Caldwell 1989). The validity and consequences of this well-established process are reflected in the adverse effects of severe and multiple defoliations on plant growth within a growing season.

An important caveat associated with plant defoliation experiments, even when conducted with field-grown plants, is that the defoliation intensities imposed are often very severe compared to actual defoliation

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patterns documented in the field. Eight of 12 defoliation studies evaluated defoliated plants at ≤6 cm, and three of these eight defoliation intensities were imposed on large tallgrass species. This suggests that while this research is valuable for understanding mechanisms of plant response to defoliation, caution should be used in translating these responses to actual grazing management applications.

Root Growth and Function. Root growth and function are increasingly suppressed with increasing intensity and frequency of defoliation because they are entirely dependent on energy derived from photosynthesis (Crider 1955). This interpretation has also been well supported by subsequent research investigating specific physiological mechanisms, including root respiration and nutrient absorption kinetics (Ryle and Powell 1975; Macduff et al. 1989). However, even though suppression of root growth following severe defoliation of individual plants is well established, the evidence that intensive defoliation suppresses root biomass within plant communities remains equivocal (Milchunas and Lauenroth 1993; McNaughton et al. 1998; Johnson and Matchett 2001). A specific mechanism has not been provided for this inconsistency, but it likely has to do with compensating root growth by less intensively grazed plants within the community or a shift in species composition to species that allocate a greater proportion of biomass belowground. Contrasting grazing responses between individual plants and communities demonstrates that caution should be used when extrapolating individual plant responses to communities and ecosystems.

Defoliation-Induced Competitive Interactions. The ability of disproportionate defoliation intensity among adjacent plants to modify intra- and interspecific competitive interactions to favor less severely grazed plants was initially proposed by Mueggler (1972). This interpretation has been substantiated with more recent and sophisticated research using isotopes of phosphorous (Caldwell et al. 1985, 1987) and nitrogen (Hendon and Briske 2002) demonstrating that both the frequency and intensity of defoliation can modify belowground competition. This series of physiological effects on competitive interactions is partially reflected in the widely observed patterns of increaser and decreaser plant species and grazing-induced changes in the species composition of plant communities.

Invalid Plant-Based Interpretations. Several well-established interpretations derived from individual plant response to defoliation have been invalidated with the advent of more sophisticated experimental procedures. This brief summary of refuted interpretations is intended not to criticize this early work, but merely to indicate that the knowledge base supporting grazing management has and will continue to advance as more research information is obtained.

Apical Dominance and Tillering. Apical dominance was promoted as the primary mechanism controlling tiller initiation following defoliation of perennial grasses. It was based on the direct hypothesis of auxin action indicating that removal of the apical meristem terminated supply of the growth inhibitor auxin to the axillary buds near the base of the tiller and thereby allowed their outgrowth into new tillers (Leopold 1949). Physiologists considered this concept invalid in the 1950s, Jameson (1963) concluded that this interpretation of apical dominance was not supported by evidence for rangeland grasses, and this conclusion was corroborated by a larger data synthesis of perennial grasses (Murphy and Briske 1992). The traditional concept of apical dominance as applied in grazing management was a partial and overly restrictive interpretation of tiller initiation in perennial grasses. A complete understanding of the mechanisms contributing to tiller initiation is yet to be developed, but it is likely a multivariable processes regulated by several interacting physiological and environmental variables (Tomlinson and O'Connor 2004).

Carbohydrate Reserves as Indicators of Regrowth. Carbohydrate reserves were proposed as an index of potential plant regrowth, and this concept was frequently applied in grazing management during the latter half of the 20th century and is still applied in limited cases. Since carbohydrate reserves decrease following plant defoliation, it was widely assumed that they must be

a major source of carbon supporting leaf regrowth (Briske and Richards 1995). A more thorough evaluation of plant carbon balance indicated that root carbohydrates were used primarily within root systems rather than being allocated aboveground to support regrowth and that reserve pools of perennial grasses contained very small amounts of carbon that contributed to regrowth for only 1–3 d before leaf photosynthesis once again became the primary carbon source (Richards and Caldwell 1985). Moreover, it appears that a consistent, positive relationship between the size of the carbon reserve pools and grass regrowth had never been established in support of this widely used interpretation (Busso et al. 1990). In retrospect, the concept of carbohydrate reserves was founded on an oversimplified interpretation of carbohydrate patterns in grasses, and it never had great relevance to grazing management. Residual leaf area and the availability of meristems, in the presence of favorable environmental conditions, are now recognized to provide more reliable indicators of plant regrowth following defoliation (Briske and Richards 1995). Ironically, emphasis on the maintenance of carbohydrate reserves in perennial grasses inadvertently applied these valid indicators of plant growth and thereby indirectly contributed to efficient grazing management.

The hierarchical structure of ecological systems describes the nested levels of ecological organization that coincide with increasing complexity and interaction among components within systems. This hierarchical structure determines why it is possible for even well-established processes at the level of individual plants to not directly translate to communities and ecosystems. For example, recall that the well-established reduction in root growth following intensive defoliation of individual plants is not consistently expressed as a reduction of root biomass within grazed communities (Milchunas and Lauenroth 1993; McNaughton et al. 1998). This inconsistent response suggests that processes and interactions within populations or communities are overriding or mitigating the negative root response of at least some of the plant species. Reductionist investigations of individual plants produce valuable mechanistic insights, but they may be too narrow in scope to identify important interactions and trade-offs at higher scales to make them relevant for direct management application (Briske 1991). Plant-based research over the past century indicates that grazing management recommendations should not be developed exclusively from processes derived at the individual plant level without at least partial verification of the anticipated response within communities or ecosystems. This is a rather sobering conclusion after nearly a century of individual plant-oriented research, but it does provide evidence of maturation and progress within the rangeland profession.

RECOMMENDATIONS

The following recommendations have emerged from our evaluation of the benefits of NRCS prescribed grazing practices with the relevant experimental literature. They are presented to enhance the effectiveness of the current conservation planning standard and to emphasize the CEAP goals addressing environmental quality of managed lands, including the assessment of multiple ecosystem services.

Priorities and Approaches to Conservation Planning

Conservation planning would benefit from a substantial shift in priorities that deemphasize the independent development of facilitating practices (e.g., fencing, roads, and pipelines) and reemphasize integration of these practices with adaptive management decisions (e.g., stocking rate, drought management, and monitoring) to promote environmental quality of rangelands as recommended by CEAP. With the clear exception of improved livestock distribution, there is no indication that facilitating practices alone directly promote effective environmental conservation. The function of grazed ecosystems is similarly controlled by several dominant environmental variables, albeit over diverse social and environmental conditions, that are expressed in dynamic forage production patterns within and among years establishing that management decisions, especially during critical periods, can have profound effects on grazed ecosystems. The environmental variables and many of the social variables cannot be directly managed, but

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recognition and planning for their occurrence with effective adaptive management plans at both the tactical and the strategic level can minimize their detrimental consequences to both production and conservation goals. Increased development and delivery of contingency planning protocols are required to effectively cope with these variable conditions common to most grazing enterprises. These tools should emphasize dynamic stocking rate determinations and provisions to support flexible management strategies, including effective destocking and restocking tactics and the potential to develop reserve forage supplies (e.g., Sharrow and Seefeldt 2006; Hanselka et al. 2009; Torell et al. 2010).

We recommend that additional decision support tools and guidelines be developed to inform adaptive grazing management decisions, especially during critical events and seasons. Current information and technology will support development of novel, comprehensive approaches for implementing dynamic stocking rate determinations that can be effectively incorporated into management plans and monitored by landowners. An undertaking of this magnitude will require investment of considerable intellectual and financial capital, but the experimental evidence strongly confirms that site-appropriate stocking rates represent the very foundation of sustainable grazing management and associated conservation benefits. These tools could target specific landowners via conservation planning or be more generally accessible through AFGC, (American Forage and Grassland Council), GLCI (Grazing Lands Conservation Initiative), SRM (Society for Range Management), or SWCS (Soil and Water Conservation Society) publications and venues or made available on NRCS websites. Incentives could be variously structured to encourage use and adoption of these tools and approaches. Conservation plans may even require participation in a set number of instructional activities to attain and maintain program eligibility.

Forage Inventory Assessment and Monitoring

Development and implementation of forage inventory and monitoring protocols in grazed ecosystems requires greater emphasis. This will require that the process of balancing forage production with animal demand be placed in the broadest possible context to include forage inventory, seasonal plant growth dynamics, and drought management over both short- and long-term periods (e.g., Sharrow and Seefeldt 2006; Hanselka et al. 2009). Static seasonal or annual stocking rates provide a broad reference, but they are insufficient to addresses wide seasonal and interannual variation in forage production common to most rangelands. Consequently, emphasis on static stocking rates results in systems being over- or understocked the majority of the time (Hart and Ashby 1998). Spatial variability of forage production, associated with variation in soils, landscape position, and local precipitation patterns, also minimizes the value of static, regional stocking rates. Use of the grazing pressure index, describing animal units per unit of forage mass over a period of time, has been recommended to standardize stocking rates and improve clarity of animal–forage relationships (Smart et al. 2010).

Stocking rates based on residual forage, determined as a percentage of site-specific annual forage productivity, minimizes the probability of over- and undergrazing at both spatial and temporal scales (Bement 1969; Clary and Leininger 2000). Management based on residual forage ensures sufficient vegetative cover to protect soils during drought and dormant seasons, enhances the capacity for plant regrowth, and provides food and cover for wildlife during stress periods. Stocking rates established to promote environmental quality on rangelands may also promote heterogeneity in structure and diversity of flora and fauna because livestock are less likely to graze uniformly across local topographic–plant community gradients within pastures.

Experimental information and available technology support development of a comprehensive approach for implementing dynamic stocking rate determinations that can be effectively incorporated into management plans with landowner participation. An undertaking of this magnitude will require investment of considerable intellectual and financial capital, but the experimental evidence directly confirm that site-appropriate stocking rates represent the very foundation of sustainable grazing management and

associated conservation benefits. Management for appropriate stocking rates not only supports conservation goals, but it also forms the basis for effective drought management strategies and sustainable long-term economic returns (Manley et al. 1997; Hart and Ashby 1998; Torell et al. 2010).

Alternative approaches are required to more directly and effectively incorporate dynamic, site-specific stocking rate assessments into overall management strategies and conservation planning. Landowner incentives could be provided to encourage adoption of forage inventory and monitoring as well as the grazing adjustments suggested by these protocols. These tools and guidelines are required to more closely estimate actual forage utilization or grazing intensity so that this information can be integrated into an adaptive management framework that emphasizes and supports flexible grazing management. Existing annual forage production curves emphasizing specific reference points that are critical to the attainment of various management and conservation goals (e.g., midpoint and end of growing season, critical wildlife requirements, and sensitivity of riparian zones) require greater attention and user friendly access. Readily accessible monthly and seasonal precipitation probabilities derived from long-term regional climatic records would also support forage inventory decisions (Andales et al. 2006). These tools may represent simple, direct measures of forage availability as well as more complicated procedures to forecast drought and forage production that could be implemented in various combinations at various temporal and spatial scales. Specific recommendations to support dynamic stocking rate determinations and promote adaptive management are summarized below.

Estimation of Residual Biomass to Determine Grazing Intensity. Estimates of residual forage could be used as a means to determine site- and period-specific stocking rates and grazing intensities, especially during drought conditions. This is a well-established management procedure that has a strong ecological basis focused on soil protection, continued surface hydrological function, and maintenance of sufficient residual plant material to provide a source of regrowth when rainfall occurs (Bement 1969; Bartolome et al. 1980; Blackburn 1984; Gifford 1985; Clary and Leininger 2000). Recommendations could be incorporated within conservation plans requesting that land managers periodically monitor residual biomass, at intervals and locations relevant to management objectives, following a prescribed set of procedures. These residual biomass records could be maintained as part of the ongoing conservation plan to support longer-term stocking rate adjustments and overall adaptive management (Bement 1969; Clary and Leininger 2000).

Forage Production and Drought Forecasting. Major technical advances have occurred in the forecasting of forage production and drought that could be used to support both tactical (within the growing season) and strategic (multiple growing seasons) grazing management decisions at regional levels. Forage production models such as GPFARM (Great Plains Framework for Agricultural Resource Management; Andales et al. 2006) could be linked with 6–14-d, 1-mo, and 3-mo precipitation and temperature forecasts through the NOAA Climate Prediction Center (http://www.cpc.noaa.gov/index.php) to provide regional projections of forage availability. Drought projections are also provided by US Drought Monitor (http://www.drought.unl.edu/DM/monitor.html) and the Vegetation Drought Response Index (http://drought.unl.edu/vegdri/VegDRI_Main.htm). This index integrates satellite-based (MODIS) observations of vegetation conditions based on NDVI, climate data, and other biophysical information, such as land cover/land use type and soil characteristics. Maps of the Vegetation Drought Response Index have been produced every 2 wk beginning in 2009 throughout the conterminous United States that deliver continuous geographic coverage over large areas, provide regional to subcounty-scale information of drought effects on vegetation, and have inherently finer spatial detail (1-km^2 resolution) than other commonly available drought indicators, such as the US Drought Monitor. Incorporation of soil water forecasts (http://www.cpc.ncep.noaa.gov/products/Soilmst_ Monitoring/US/Soilmst/Soilmst.shtml) could further promote the accuracy of these forage production projections. Forage projections could be developed for

> “The vast majority of experimental results indicate that there is no clear advantage of any one grazing system over another in terms of ecological benefits.”

The importance of effective tactical and strategic decisions to successful grazing management is widely acknowledged, but only poorly documented. (Photo: Alexander Smart)

specific periods of management interest and provide probabilities for forage responses to dry, average, and wet conditions to ascertain various levels of management risk. Forecast information could interface with existing forage production curves previously developed by the NRCS to generate various forage inventory projections to inform management planning.

Drought Contingency Planning. It is essential that monitoring protocols be linked to drought contingency planning and management actions. It is widely recognized that the commonly employed strategy of "optimistic inaction" regarding stocking rate adjustments in response to developing drought is a major contributor to long-term rangeland degradation (Stafford Smith and Foran 1992; Thurow and Taylor 1999; Torell et al. 2010). However, it is irresponsible to delay or fail to implement drought contingency planning based on the unpredictability of drought given its frequent occurrence on most rangelands (Thurow and Taylor 1999). Renewed emphasis on drought contingency planning must integrate both economic and ecological considerations to effectively encourage managers to adopt and implement destocking options in relation to drought.

Conservative stocking rates and the formation of reserve forage or grass banks are well-established strategies for contending with economic and environmental aversion to drought risk (Thurow and Taylor 1999). During normal or wet years, these grass banks could serve as restoration programs to support prescribed burning or to promote critical ecosystem services (i.e., biodiversity and carbon sequestration). Flexible stocking is also an effective means to cope with variable precipitation and forage production (Stafford Smith and Foran 1992; Torell et al. 2010). Cow-calf herds should represent only a conservative component of total livestock holdings because of the high cost of adjusting cow numbers relative to the potential for short-term gain. Equal forage allocation to cow-calf and stockers has been recommended for ranching operations in the western United States (Torell et al. 2010). It is important to recognize that flexible stocking conveys additional costs and financial risks that will require specific decision-making tools to expand its adoption, and it may not be appropriate for risk-averse managers (Tanaka et al., this volume).

The Role of Grazing Systems

It is extremely difficult to experimentally mimic livestock movements and defoliation patterns associated with various applications of grazing strategies used by managers. However, grazing systems research has carefully evaluated the ecological responses of individual plants and communities, including wildlife populations, soils and surface soil hydrology, and their feedbacks on livestock performance, including forage intake and weight gain per animal and per unit area. These major ecological variables integrate numerous ecosystem processes sufficiently well to provide reliable guidance for the implementation and evaluation of the ecological consequences associated with grazing systems. The vast majority of experimental results indicate that there is no clear advantage of any one grazing system over another in terms of ecological benefits. Conclusions derived from these experimental data provide a sufficient basis to establish ecological guidelines for the evaluation and application of grazing systems in conservation planning and ecosystem assessment. These data directly corroborate the long-standing conclusions that weather

variability and stocking rate account for the majority of variation associated with plant and animal production and species composition changes on rangelands (Heitschmidt and Taylor 1991; Holechek et al. 2001; Derner and Hart 2007). This interpretation further emphasizes the importance of effective adaptive management to the successful operation of grazed ecosystems, including the establishment of clear goals, monitoring of resource conditions, and the ability to make appropriate and timely management adjustments. Stated in another way, there is no indication that grazing systems possess unique properties that enable them to compensate for poor management (Briske et al. 2008).

This interpretation also emphasizes that it is not sufficient to evaluate only whether grazing management is effective; we also need to determine why it is effective. This information is essential to guide development of effective conservation practices by determining whether emphasis should be focused on facilitating practices or on adaptive management skills. Although largely undocumented, the importance of effective adaptive management to successful grazing management is widely acknowledged, and it requires much greater emphasis than it has received (Stuth 1991; Brunson and Burritt 2009; Hanselka et al. 2009). Both research and monitoring are required on ranch-scale operations to more clearly evaluate the contribution of adaptive management to the success of conservation practices and to investigate the interaction between adaptive management and various grazing systems at the ranch level (e.g., Jacobo et al. 2006).

Deferment and Rest

Few evidence-based conclusions can be drawn regarding the appropriate season for grazing deferment and the benefits of long-term rest. This is partially illustrated by the inconsistent vegetation responses associated with the application of rest-rotation systems (Holechek et al. 2001). Minimal advantages may have resulted because one season of complete rest may not have been sufficient to compensate for more intensive use of grazed pastures in previous years. Vegetation responses to season of grazing and deferment are highly dependent on 1) the timing and amount of precipitation received during the growing season, 2) the intensity of defoliation, and 3) the opportunity for regrowth following defoliation. Research is required to quantify the benefits of long-term rest (> 1 yr) and alternating seasons of pasture use in successive years. Limited evidence suggests that exclusion of livestock is not necessary for recovery from moderate drought on well-conditioned rangeland (Heitschmidt et al. 2005; Gillen and Sims 2006), but it may be beneficial following severe drought that has induced substantial tiller and plant mortality (Dalgleish and Hartnett 2006; Yahdjian et al. 2006). Plants subject to light and moderate grazing often show less drought-induced mortality than plants that have been severely grazed prior to drought (Albertson et al. 1957).

Grazing can potentially be used as a tool to manage wildlife populations, and this may be particularly true when season of grazing or deferment of grazing is used to meet specific wildlife goals. Seasonal livestock use may especially benefit wildlife where only part of the range is desirable wildlife habitat and social avoidance or seasonal migration are important considerations, facilitation through improved forage quality has been demonstrated, or specific nesting requirements are an issue. In these cases livestock grazing may be imposed or deferred, depending on cover and foraging requirements of specific wildlife species. For example, some waterfowl or some upland game species require dense nesting cover, whereas some birds, such as mountain plover or curlews, choose nesting sites with very little cover and will not nest in ungrazed or lightly grazed habitat. Successful use of seasonal and deferred grazing may also be possible when pastures with limited wildlife value are available to minimize livestock use in adjacent pastures that contain critical wildlife habitat.

Stronger Linkages between Science and Management

NRCS Conservation Practice Standards should be routinely informed by both scientific and management knowledge external to the agency to ensure that the most current and vetted information available is incorporated into the conservation planning process. This represents a formidable challenge because science and management are not directly comparable endeavors (Provenza 1991), and this may partially explain why

> "The knowledge gaps identified in this synthesis need to be at least partial addressed to promote the development and adoption of more effective conservation practices in grazed ecosystems."

The diverse ecosystem services originating from rangelands require greater recognition and valuation. (Photo: USDA: Gary Kramer)

stronger science–management linkages have not been forged in the rangeland profession. Experimental research has focused on specific aspects of grazing management, including stocking rate, grazing system, and livestock distribution, in a static and independent manner, rather than on their dynamic interaction within adaptively managed ecosystems. The critical but poorly defined contribution of adaptive management to grazed ecosystems is a major impediment to the development of linkages between research and management because decision making is often excluded from experimental research even though it is central to grazing management (Briske et al. 2008; Brunson and Burritt 2009). Research requires systematic collection of information to document outcomes of various grazing strategies, while the outcomes of conservation practices standards are seldom monitored and documented. This often results in the difficult task of comparing quantitative research results with qualitative and often anecdotal management information. New organizational structures are needed to bridge the gap between research and management to support and incentivize a more comprehensive framework for conservation planning (Boyd and Svejcar 2009; Svejcar and Havstad 2009). The NRCS may wish to adopt a more formal research–management framework to address conservation programming that could be convened each time a conservation practice standard undergoes reevaluation.

Substantial differences between rangeland science and management have presented barriers to their integration throughout the history of the rangeland profession. The extensive

synthesis of experimental information provided in this document and the science–management partnership forged by this 3.5-yr undertaking represents an important initial step in attaining this goal. Greater integration and information exchange among researchers and managers would create a "win–win" situation for the profession by facilitating development of evidence-based conservation practices. This represents a necessary step if Conservation Practice Standards are to effectively adopt CEAP recommendations to provide regular assessments of the societal benefits of taxpayer investments in conservation practices. It would also enable the management community to play a more direct role in establishing the rangeland research agenda, as suggested in the following section. Effective monitoring of conservation practice outcomes will be crucial for enhancement of science–management linkages by providing a quantitative source of information exchange between these two groups.

KNOWLEDGE GAPS

The following knowledge gaps were identified in the process of summarizing and interpreting the experimental literature associated with prescribed grazing. It is anticipated that by highlighting these poorly understood issues, they may receive additional research attention and funding to promote greater understanding. It is critical that these knowledge gaps be at least partially addressed to promote the development and adoption of more effective conservation practices in grazed ecosystems.

Ecosystem Processes and Services in Grazed Ecosystems

Traditionally, grazing research has focused on several ecological variables, including plant and animal production and, to a lesser extent, patterns of species composition change and wildlife responses and habitat. These variables provide a valuable, but admittedly narrow foundation on which to assess ecosystem services and environmental quality in grazed ecosystems. Research programs designed to increase our understanding of ecosystem processes and the provisioning of ecosystem services are desperately needed. Relevant topics include plant functional groups, soil health and sustainability, biodiversity, carbon sequestration, greenhouse gas emissions, drought and drought recovery, and spatial heterogeneity of ecosystem and landscape structure.

Ecosystem Restoration and Conservation Strategies

Even though grazing management was initiated to halt and reverse the adverse effects of overgrazing on rangeland ecosystems, restoration of grazed ecosystems has received limited research attention in the past several decades. Research has been focused primarily on optimization of livestock production during the past 30 yr with use of intensified grazing systems. Consequently, experimental information regarding the season of utilization or deferment that is most appropriate to restore degraded ecosystems or to promote various conservation strategies is limited. Research addressing individual bunchgrass responses to defoliation in the field indicates that mid-growing season is the most sensitive period for defoliation. However, we are unaware of community-level field studies that corroborate this conclusion. Similarly, individual plant research has imposed very severe defoliation intensities compared to observed utilization rates in grazed ecosystems so that the direct application of these results to management is limited. Plant, community, and ecosystem responses to realistic grazing patterns would benefit from further documentation.

Contributions of Adaptive Management

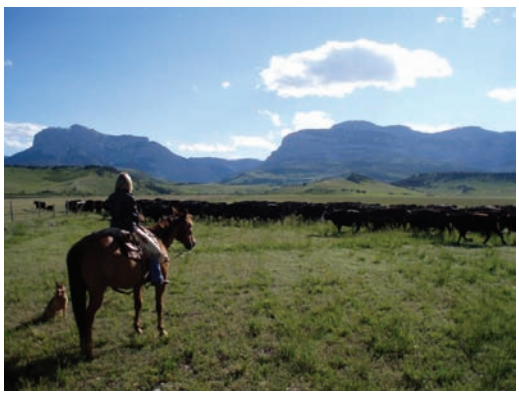
Management goals, abilities, and opportunities as well as personal goals and values (e.g., human dimensions) are inextricably integrated within grazing management, and they are likely to interact with the adoption and operation of grazing systems to an equal or greater extent than the underlying ecological processes (Briske et al. 2008). Therefore, research and monitoring approaches need to explicitly document the contribution of adaptive management within ecosystems to promote a more comprehensive understanding of successful grazing management (Brunson and Burritt 2009; Budd and Thorpe 2009). The potential synergistic effects of grazing systems and adaptive management inputs have not been examined experimentally at the level of the ranch enterprise (Briske et al. 2008; Brunson

"Research and monitoring approaches need to explicitly document the contribution of adaptive management within ecosystems to promote a more comprehensive understanding of successful grazing management."

and Burritt 2009). Successful research in this area will require direct involvement of social and political scientists addressing these critical human dimensions issues and their interactions with ecological systems. A novel experimental approach used by Jacobo et al. (2006) compared adjacent ranches that had employed unique grazing systems to achieve the optimal production outcome. The strength of this approach is that it enables researchers to evaluate outcomes reflecting the entire ranch enterprise, including the capacity to adaptively manage for the best possible outcomes, within the context of the respective grazing system. This approach simultaneously evaluates ecological and managerial responses, but it has yet to be determined whether it will be possible to distinguish between these two responses. Similarly, incentives and barriers of various social institutions influencing the adoption of conservation practices have received minimal research emphasis given their importance to the management of complex adaptive systems (Stafford Smith et al. 2007).

Evaluation of Large-Scale Ecosystem Responses

Grazing research has not adequately assessed the effects of grazing at large scales (Bailey et al. 1996; Archibald et al. 2005), which often demonstrate the occurrence of patch- and area-specific grazing. Smaller experimental pastures usually result in more uniform distribution of grazing intensity, which may not appropriately describe how domestic grazing animals utilize large landscapes or, in the case of native ungulates, how they migrate regionally. The direct application of research results obtained in small-scale experiments (< 200 ha) to large ranch enterprises may not be entirely appropriate because the ecological processes of interest often do not scale in a linear fashion (Fuhlendorf and Smeins 1999; Peters et al. 2006). Investigations of the potential benefits of grazing systems at large scales require further evaluation, and the evaluation metrics should involve a variety of ecosystem services, such as firm-level production, biodiversity concerns, watershed function, and wildlife habitat.



Current NRCS grazing practices are appropriate in many respects, but multiple opportunities exist to improve their effectiveness. (Photo: Sonja Smith)

Integration of Complex Ecosystem Components

The complexity of grazed ecosystems resides in the broad array of interacting variables associated with both ecological and human systems. A wide range of ecological variation is associated with rainfall regime (i.e., amount, seasonality, and intra- and interannual variability), vegetation structure, composition, and productivity and soils, prior land use, and livestock characteristics (i.e., breeds, prior conditioning, and previous experience). This tremendous ecological variability is paralleled by large, but unappreciated variability associated with the commitment, ability, goals, and opportunities of managers and associated stakeholders dependent on the services of these ecosystems (Briske et al. 2008; Brunson and Burritt 2009). The success and benefits that accrue from conservation practices within these complex systems is dependent on three unique activities. First, the conservation practices must be based on sound managerial and ecological principles; second, practices must be effectively incorporated into the overall conservation plan; and, third, they must be appropriately applied, maintained, and monitored by ecosystem managers. The third component addressing manager or landowner commitment and capability is most widely overlooked and can be addressed only from a human dimensions perspective.

A robust ecosystem management framework capable of accommodating both ecological processes and human activities, as well as their interactions, is required to conceptualize, interpret and manage complex adaptive systems characteristic of rangelands. This will require the development of an information base that consists of local knowledge, management and policy experience, and science-based information mediated through an adaptive institutional framework

(Herrick and Sarukhan 2007; Reynolds et al. 2007). This framework must be coupled with current and emerging technologies to provide estimates of remotely sensed data to address multiple feedbacks between the social and ecological components at several scales. Ecological site descriptions may provide the platform on which to integrate these sources of information, but the rangeland profession is lacking an institutional structure to house and coordinate relevant ecosystem components and processes at landscape and regional scales (Bestelmeyer et al., this volume). Approaches that involve integration of ecological scales and human dimensions, coupled with effective monitoring protocols capable of evaluating both ecological and social metrics, will likely drive the next major advance in effective rangeland stewardship.

CONCLUSIONS

An extensive evaluation of the published experimental evidence relevant to grazing management broadly supports the overall USDA-NRCS approach to prescribed grazing and validates the ecological foundations of many of the purposes addressed in this conservation practice standard. The equivocal nature of a portion of these findings is a consequence of experimental research and conservation planning pursuing different objectives, with unique approaches that are often conducted at different scales and an unfortunate legacy of minimal interaction between science and management within the rangeland profession. Nevertheless, inferences drawn from these experimental data indicate that the NRCS conservation purposes addressing prescribed grazing can potentially be realized, if implemented appropriately, as indicated by the ability for grazing management practices to affect all seven stated conservation purposes. The challenge of grazing management is establishing the appropriate relationships between various management practices and the intended purposes or outcomes, in diverse environmental and social conditions, especially when multiple and often competing purposes are involved.

The experimental data unequivocally document that stocking rate, coupled with effective livestock distribution, is the single most important management variable influencing production and conservation goals in grazed ecosystems. Therefore, guidelines, tools and incentives that promote appropriate management decisions have the potential to enhance the effectiveness of conservation outcomes and increase the cost–benefit ratio of conservation investments. Guidelines promoting the goal of balancing forage production with animal demand should be placed in the broadest possible context to include forage inventory, seasonal plant growth dynamics, and drought management over both the short and the long term. Existing annual forage production curves emphasizing specific reference points that are critical to the attainment of various management and conservation goals, supported by monthly and seasonal precipitation probabilities, require greater emphasis and user-friendly access to support forage inventory decisions. The adoption of this approach will require a major shift in NRCS programmatic emphasis from those promoting facilitating practices in the form of infrastructure development to those promoting timely and effective adaptive management actions.

Experimental evidence indicates that grazing systems, in the absence of adaptive management, explain little additional variability beyond that of stocking rate and weather variation for the variables of plant and animal production, species composition of plant communities, soil surface hydrological function, and wildlife populations. In addition, the major assumptions on which short-duration rotational grazing is partially based, including greater control over grazing patterns, minimization of multiple defoliations within individual grazing periods, and greater forage quality, have received only equivocal experimental support. Current evidence suggests that implementation of grazing systems, without incorporation of the additional elements of prescribed grazing, is insufficient to address the array of complex and dynamic conditions inherent to grazed ecosystems. However, the potential contributions of grazing systems to broader conservation goals and ecosystem services, at landscape or regional scales, and their potential

> “CEAP provides an excellent platform to promote greater science–management integration by bringing together researchers and NRCS personnel for an evidence-based assessment of Conservation Practice Standards”

interactions with adaptive management have yet to be evaluated.

Several important knowledge gaps have been identified in the experimental literature associated with prescribed grazing. These include 1) grazing effects on ecosystem services, 2) ecosystem restoration and conservation strategies, 3) contributions of adaptive management actions, 4) evaluation of larger-scale ecosystem responses, and 5) integration of information within complex ecosystems. The greatest deficiency encountered in this evaluation of supporting experimental data was the paucity of information documenting the impact of adaptive management on grazing management effectiveness and conservation outcomes. It is critical that these knowledge gaps be at least partially resolved in the near future to promote further advances in the ecology and management of grazed ecosystems.

The overarching conclusion of this assessment is that even though the current conservation practices for prescribed grazing are appropriate in many respects, reorganization to implement three major modifications would greatly increase their effectiveness. First, greater emphasis should be placed on programs to support management skills and management effectiveness beyond that of financial incentives supporting the independent development of infrastructure. There is no clear indication that installation of facilitating practices in the form of water developments and fencing directly contribute to conservation benefits in the absence effective management. Second, a system of regular and frequent monitoring needs to be incorporated into conservation planning to directly assess both the short-term and the long-term benefits derived from conservation practices. Monitoring information will directly support adaptive management to optimize conservation outcomes per unit investments and document the ecological benefits of conservation practices on the nation's rangelands as recommended by CEAP. Third, incorporate the intent and recommendations of CEAP by focusing on environmental quality, ecosystem services, and societal benefits associated with prescribed grazing in addition to sustainable production outcomes (e.g., Dunn et al. 2010).

Revisions to this conservation practice standard should be informed by both scientific and management knowledge external to the agency to ensure that the most current and vetted information available is incorporated into conservation planning and assessment procedures. CEAP provides an excellent platform to promote greater science–management integration by bringing together researchers and NRCS personnel, as well as other stakeholders, for an evidence-based assessment of Conservation Practice Standards and approaches to conservation planning. We recommend that this integrated science–management team approach should be formalized in the agency and used to revise Conservation Practice Standards and set priorities and goals for conservation planning.

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APPENDIX I A.

Published experiments used to evaluate plant and animal production to continuous grazing (CG) and rotational grazing (RG) at (a) equal stocking rates and (b) higher stocking rates for rotational grazing. See Figure 3 for a graphical presentation of the comparative results (modified from Briske et al. 2008).

Study	Location	Ecosystem	Length (yr)	Grazing system	
Stocking rate equal for rotational and continuous grazing					
McCollum et al. (1999)	Oklahoma	Tallgrass prairie	5	Short-duration grazing (SDG)	
Gillen et al. (1998)	Oklahoma	Tallgrass prairie	5	SDG	
Cassels et al. (1995)	Oklahoma	Tallgrass prairie	5	SDG	
Owensby et al. (1973)	Kansas	Tallgrass prairie	17	Deferred rotation	
Wood and Blackburn (1984)		Southern mixed-grass prairie	5	High-intensity/low-frequency and deferred rotation	
Kothmann et al. (1971)	Texas	Southern mixed-grass prairie	8	Merrill	
Merrill (1954)	Texas	Southern mixed-grass prairie	4	Merrill	
Fisher and Marion (1951)	Texas	Southern mixed-grass prairie	8	Rotational	
McIlvain and Savage (1951)	Oklahoma	Southern mixed-grass prairie	9	Rotational	
Derner and Hart (2007a)	Wyoming	Northern mixed-grass prairie	25	SDG	
Manley et al. (1997)	Wyoming	Northern mixed-grass prairie	13	SDG and deferred rotation	
Biondini and Manske (1996)	North Dakota	Northern mixed-grass prairie	6	SDG	
Hart et al. (1993)	Wyoming	Northern mixed-grass prairie	5	SDG	
Hepworth et al. (1991)	Wyoming	Northern mixed-grass prairie	4	SDG and deferred rotation	
Hart et al. (1988)	Wyoming	Northern mixed-grass prairie	6	SDG and deferred rotation	
Rogler (1951)	North Dakota	Northern mixed-grass prairie	25	Deferred rotation	
Derner and Hart (2007b)	Colorado	Shortgrass prairie	9	SDG	
Smoliak (1960)	Alberta, Canada	Shortgrass prairie	9	Deferred rotation	
Hubbard (1951)	Alberta, Canada	Shortgrass prairie	6	Deferred rotation	
Laycock and Conrad (1981)	Utah	Sagebrush–grassland	7	Rest-rotation	
Hyder and Sawyer (1951)	Oregon	Sagebrush–grassland	11	Rotational	
Holechek et al. (1987)	Oregon	Mountain rangeland	5	Rest-rotation and deferred rotation	
Murray and Klemmedson (1968)	Idaho	Cheatgrass (*Bromus tectorum*) dominated	3	Seasonal rotation	
Martin and Severson (1988)	Arizona	Grass–shrub complex	13	Santa Rita, 1-herd, 3-pasture, 3-yr rotation	
Martin and Ward (1976)	Arizona	Desert grassland	7	Alternate-year seasonal rest	
Winder and Beck (1990)	New Mexico	Semidesert	17	3-pasture rotation	
Gutman et al. (1990)	Israel	Mediterranean grassland	2	Rotational	
Gutman and Seligman (1979)	Israel	Mediterranean Foothill Range	10	Rotational	
Ratliff (1986)	California	Annual grassland	8	Rotational	

	No. of paddocks in rotation	Size of paddocks (ha)	Plant production/standing crop	Livestock production per head	Livestock production per land area
	8	1.8–3.3		CG > RG	CG > RG
	8	1.8–3.3	ND		
	8	1.8–3.3	ND at peak standing crop RG > CG at end of grazing season		
	3	24	RG > CG	CG > RG	CG > RG
	4	120	ND		
	4	?		RG > CG	RG > CG
	4	24		ND	ND
	5	4		ND	ND
	3	6.7–10		ND	ND
	8	1–2	ND		
	4–8	1–3	ND	ND	ND
	6	32	ND		
	8	24		ND	ND
	4–8	1–3		ND	ND
	4–8	1–3	ND	ND	ND
	3	9.4		RG > CG	RG > CG
	7	65		ND	ND
	2	61	ND	CG > RG	CG > RG
	3	27–40		ND	ND
	3	447–777	ND	ND	ND
	3	850		CG > RG	CG > RG
	2	57–67		ND	ND
	4	?	ND	ND	ND
	3	308–1 979	ND		
	24	0.004	ND		
	3	?		ND	ND
	6	25.5–33.0	ND	CG > RG	CG > RG
	3	25.5–33.8	ND	ND	ND
	3	30	ND	CG > RG	CG > RG

APPENDIX I A. *continued*

Study	Location	Ecosystem	Length (yr)	Grazing system
Stocking rate equal for rotational and continuous grazing				
Heady (1961)	California	Annual grassland	5	Deferred rotation
Barnes and Denny (1991)	Zimbabwe	Veld	6	SDG
Fourie and Engels (1986)	South Africa	Veld	4	SDG
Fourie et al. (1985)	South Africa	Veld	4	SDG
Kreuter et al. (1984)	South Africa	Veld	3	SDG
Walker and Scott (1968)	Tanzania		2	Rotational
Bogdan and Kidner (1967)	Kenya	Woodland–grassland	5	Rotational, deferred rotational
Higher stocking rate for rotational grazing				
Jacobo et al. (2000)	Argentina	Temperate grasslands	3	SDG
Heitschmidt et al. (1987)	Texas	Southern mixed-grass prairie	4	SDG
Heitschmidt et al. (1982a)	Texas	Southern mixed-grass prairie	2	SDG
Heitschmidt et al. (1982b)	Texas	Southern mixed-grass prairie	19	Merrill
Reardon and Merrill (1976)	Texas	Southern mixed-grass prairie	20	Deferred rotation
Hirschfeld et al. (1996)	North Dakota	Northern mixed-grass prairie	2	SDG
Volesky et al. (1990)	South Dakota	Northern mixed-grass prairie	2	SDG
Kirby et al. (1986)	North Dakota	Northern mixed-grass prairie	2	SDG
White et al. (1991)	New Mexico	Blue grama	6	SDG
Pitts and Bryant (1987)	Texas	Shortgrass prairie	4	SDG
Anderson (1988)	New Mexico	Tobosa (*Hilaria mutica*) dominated	2	SDG

APPENDIX I B. Literature cited in the construction of Appendix I A above.

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	No. of paddocks in rotation	Size of paddocks (ha)	Plant production/standing crop	Livestock production per head	Livestock production per land area
	3	5.4	ND	CG > RG	CG > RG
	4–8	?		ND	ND
	6	5		CG > RG	CG > RG
	6	5	ND		
	6	?		CG > RG	CG > RG
	3	5		CG > RG	CG > RG
	3–4	0.6–0.8		CG = RG	CG = RG
	10–12	45	ND		
	16	33	CG > RG		
	10	4		ND	RG > CG
	4	?		ND	RG > CG
	4	?	ND		
	8	16.25	ND		
	16	2.2		CG > RG	RG > CG
	8	16	RG > CG		
	9	45–210	ND		
	16	3	ND	ND	ND
	10	3.5	ND	CG > RG	CG > RG

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CHAPTER

2

Assessment of Prescribed Fire as a Conservation Practice

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“A number of studies across US rangelands reported that shrubs and trees increase and herbaceous vegetation decreases with long-term fire removal, so maintaining or increasing relative abundance of herbaceous vegetation requires periodic fire.”

Assessment of Prescribed Fire as a Conservation Practice

2

Samuel D. Fuhlendorf, Ryan F. Limb, David M. Engle, and Richard F. Miller

INTRODUCTION

Fire has played a key role in the formation of most rangeland ecosystems in North America (Axelrod 1985) and the world (Bond et al. 2003; Keeley and Rundel 2005). Alteration of fire regimes on US rangelands since European settlement has created cases of severely altered ecosystems that can eventually result in no-analog, novel, or emerging ecosystems (House et al. 2003; Hobbs et al. 2009). According to the Landscape Fire and Resource Management Planning Tools Project (LANDFIRE; an interagency vegetation, fire, and fuel characteristics mapping program sponsored by both the US Department of Interior and the US Department of Agriculture [USDA]–Forest Service), three-fourths of US lands dominated by native vegetation show moderate or high departure from reference conditions as a result of altered fire regimes (The Nature Conservancy 2009). Because most rangelands are considered fire-dependent ecosystems, restoring historical fire regimes is fundamentally important when the management goal is to restore or maintain the potential (or historical) natural communities. For most ecological sites, the historical plant community was maintained by fire, and removing fire will cause the community to cross a threshold, often to woody plant dominance with reduced livestock production and loss of other ecosystem services. Rapid and extensive woodland expansion on rangelands clearly reflects the essential role of fire in the maintenance of historical rangeland ecosystems. These recent changes in land cover patterns emphasize that restoration of historical fire regimes are necessary to maintain these historical communities as outlined in the Natural Resources Conservation Service (NRCS) Ecological Site Descriptions. Yet, the implementation of prescribed burning as a conservation practice has been overshadowed by the implementation of other practices, especially prescribed grazing. In Oklahoma for example, from 2004 to 2008, NRCS implemented prescribed burning on 84 700 ha compared to 919 800 ha for prescribed grazing during 2004–2008. This 10-fold difference in the application of these two conservation practices clearly identifies the higher priority placed on grazing compared to that of burning. Considering that NRCS grazing practices operate over multiple years and that the practice of prescribed burning is a one-time application, the effective difference is actually considerably larger than 10-fold. Disproportionate implementation of these two practices is influenced by the complexity of social interactions among agencies, the general public, and public policy. Social and policy concerns are extremely different across various rangeland regions, ranging from complete acceptance of fire cultures (e.g., Flint Hills of Kansas and Oklahoma) to attempts to completely remove fire from the landscape (e.g., Great Basin).

With a few exceptions, fire regimes have been altered through intentional fire suppression and by grazing that uniformly reduces fuel loads. Therefore, invasion of woody plants (both native and nonnative) has converted many shrublands and grasslands to forests or woodlands because of the absence of fire for abnormally long periods after European settlement. In contrast, other rangelands, notably those of the Great Basin, have been largely invaded by exotic herbaceous species that increase fine-scale fuel homogeneity, which greatly alters the fire regime by increasing fire frequency. State-and-transition models suggest conversions to woody plant dominance and exotic annuals can eventually become irreversible and result in alternative stable states. Although rangeland management

A prescribed fire in a mixed grass prairie that has been invaded by woody plants. (Photo: John Weir)

Suppression of fire allows invasion of woody plants into grassland habitats. Prior to European settlement, First Nations actively burned the landscape for many reasons and maintenance of those ecosystems requires periodic fires. (Photo: Sam Fuhlendorf)

professionals generally support using fire in rangeland ecosystems, a long history of exclusion, uncertainty about the effects of fire, increased wildland–urban interface, socioeconomics, and natural resource policy are formidable barriers to reintroducing fire except in those ecosystems in which the fire return interval has been shortened. However, as long as maintaining ecosystem structure within a historical context is emphasized, fire regimes must be restored across most rangelands.

The USDA-NRCS Practice Standard for Prescribed Burning (CODE 338) describes the following purposes:

1. to control undesirable vegetation;
2. prepare sites for harvesting, planting, and seeding;
3. to control plant diseases;
4. to reduce wildfire hazards;
5. to improve wildlife habitat;
6. to improve plant production quantity and/or quality;
7. to remove slash and debris;
8. to enhance seed production;
9. to facilitate distribution of grazing and browsing animals; and
10. to restore and maintain ecological sites.

The Conservation Effects Assessment Program (CEAP) was initiated to determine the extent to which experimental data present in peer-reviewed research literature support these purposes. The general and value-laden nature of these purposes makes them extremely difficult to assess against experimental data; therefore, we analyzed the research literature to establish the ecological effects of prescribed fire from a broader perspective. Specifically, we evaluated the research literature available on plants, soil, water, wildlife, arthropods, livestock,

fire management, fire behavior, smoke management, socioeconomics, air quality, fire history, and human health. These topics were selected based on input from rangeland CEAP teams focused on other conservation practices and initial evaluation of the literature in terms of topics that were covered sufficiently to draw meaningful conclusions. We also addressed issues related to spatial scale, temporal scale, and other general descriptions of the body of research and we then related our findings to the specific NRCS purposes for the practice of prescribed burning.

DEFINING OUR LITERATURE DATABASE

Evaluation of the peer-reviewed literature on prescribed fire first required determining methods to query the entire body of scientific literature on the topic. We wanted to include all relevant papers, but we limited the scope of the search to minimize less-relevant papers. We searched for papers that focused on fire (preferably prescribed), but largely excluded fire research from forested systems, which dominates the fire research literature. Many papers that report fire research on rangelands do not include the term *prescribed*, and many relevant papers do not use the term *rangeland.* We used multiple approaches (Table 1) to identify the most acceptable body of literature to evaluate. The data set built from the search with the term *prescribed fire* omitted numerous important papers from the pool, and many of the papers included some discussion of fire but with minimal or no data related to fire. Therefore, our final search used the term *fire*, which also located articles with *prescribed fire* in the title, to broaden the search. Although this approach excluded papers that reported research from regionally important ecosystem types (e.g., shinnery oak or chaparral vs. shrubland) and papers in which the title contained other key fire-related words (e.g., burned, burning, and prescribed burning) but not fire, the search located more than 500 papers (Table 2). Assuming our search provided an adequate, unbiased sample of the literature, we evaluated the search database to determine the nature of information available through the peer-reviewed literature. We then supplemented this information with papers that addressed specific topics within our charge for this project. As with the comprehensive search, we used Web of Science to search for papers on a particular topic. We justified limiting our search to the indexed literature on the basis that it is widely accepted as scientifically valid and the primary science published in peer-reviewed literature.

TABLE 1. Number of papers identified for six topics in a Web of Science search of peer-reviewed journals. Each number represents the number of papers from the Web of Science for each topic listed.

Topic	Fire	Prescribed Fire
Rangeland	172	48
Shrubland	265	24
Grassland	931	138
Grazing	831	95
Wildland	494	83
Forest	6 648	671
Total with forest	9 341	1 059
Total without forest	2 245	318

TABLE 2. Numbers of papers identified in a Web of Science search using fire (not prescribed fire) and each of the words in the first column in the title of the paper. These papers formed the initial database that was analyzed.

Title search combining fire and one of the following words	Number of papers
Shrubland	24
Savanna	157
Grazing	86
Woodland	61
Wildland	150
Rangeland	18
Grassland	107
Total	563

EVALUATION OF THE DATA SET

Of the 563 papers (available in January 2008) from our query through Web of Science (Table 2), 474 papers were accessible and confirmed to be peer-reviewed research papers. Of these 474 papers, less than 10 papers were published annually from 1967 through 1989 followed by a marked increase with 20 or more papers

FIGURE 1. The number of papers published per year from a total of 563 papers published on rangeland fire. See text and Table 1 for explanation of papers selected.

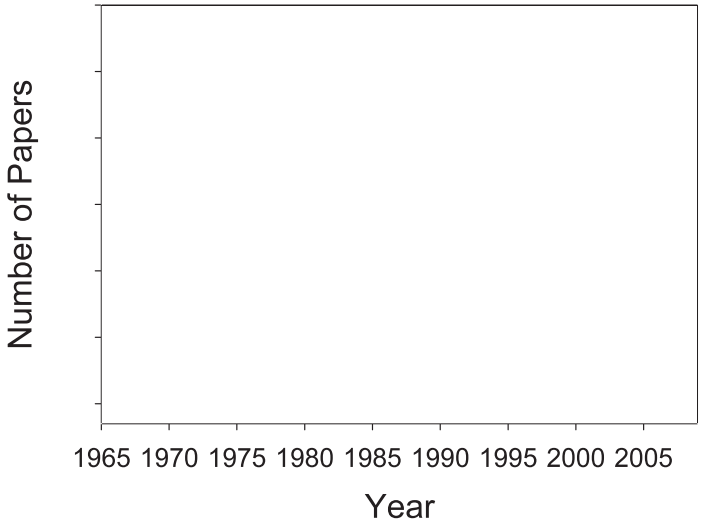


TABLE 3. Number of papers published by peer-reviewed scientific journals between1967 and 2007 based on a Web of Science search.

Journal	Number of papers
International Journal of Wildland Fire	25
JRM/Rangeland Ecology and Management	23
Ecology	15
Journal of Applied Ecology	15
Forest Ecology and Management	13
Vegetatio /Journal of Vegetation Science	15
Journal of Ecology	12
Journal of Tropical Ecology	12
African Journal of Ecology	11
Austral Ecology	10
Plant Ecology	10
Journal of Arid Environments	8
Remote Sensing of Environment	8
Australian Journal of Ecology	7
Biotropica	7
Ecological Modelling	7
Journal of Biogeography	7

published each year from 2000 to 2007 (Fig. 1). These data suggest that the research community may view fire with increasing importance but these conclusions could be limited by the words included in the search.

An important outcome of the search was that rangeland fire research literature is dispersed among numerous ecological journals. Furthermore, most continents are well represented in the research, and topics include those not explicitly addressed in the NRCS purposes for prescribed burning. More than 150 journals, mostly international ecological or applied ecological journals, published rangeland fire research (Table 3). Most of the research was located in North America, but substantial research was conducted in Africa, Australia, South America, and Europe. Research from the United States contributed 214 of the 474 papers in the data set. The majority of papers reported research on plants, fire management, soils, fire behavior, socioeconomics, and wildlife. Authors described their papers as addressing a variety of vegetation types, with over half of the papers classified as savannas and grasslands (Fig. 2). Most of the articles recognized by our search terms in the United States reported research from the Great Plains, followed by the West Coast, Intermountain West, Eastern Forests–Grasslands, and Desert Southwest (Table 4). Topics in the database focused on plants, socioeconomics, fire management, soils, fire behavior, and wildlife, in respective order from highest to lowest, with all other topics having 10 or fewer papers (Table 4).

Perhaps the most revealing outcome of our search was that it uncovered critical limitations to applying the research literature to management applications, which is a fundamental barrier to constructing research-informed purposes for prescribed burning. First, we found that most of the research was conducted at temporal and spatial scales inappropriate to management. Second, the fire research literature generally ignores fire as a dynamic disturbance process that varies in frequency, intensity, and time since fire (most studies are less than 5 yr postfire). Finally, most research failed to evaluate fire in the context of other disturbances, such as grazing and drought, on complex landscapes. More than

half of the papers reported research conducted on experimental units of 1 ha or less and many studies were conducted on much smaller plots and on individual plants (Fig. 3). Fifteen percent of the studies were based on modeling and 6% on geographic information systems with minimal field evaluations or immediate application to management. Many studies (27%) simply compared a burn treatment to an unburned control, which obviously simplifies fire to the point of irrelevance to management.

Fire regime, the features that characterize fire as a disturbance within an ecosystem—fire frequency, severity, behavior (i.e., fire intensity), predictability, size, seasonality, and spatial pattern (Morgan et al. 2001)—was rarely evaluated. Only 12.5% of the papers focused on *fire frequency*, and only 4% focused on understanding changes that occur over variable times since fire. *Fire season* was evaluated in 7% of the studies and *fire intensity* was evaluated in 9% of the studies. Most studies failed to specifically discuss the interaction of fire with

FIGURE 2. Percentage of 474 rangeland fire studies within each of seven major vegetation types as described by the authors. Papers were published in peer-reviewed journals and located in a Web of Science search.

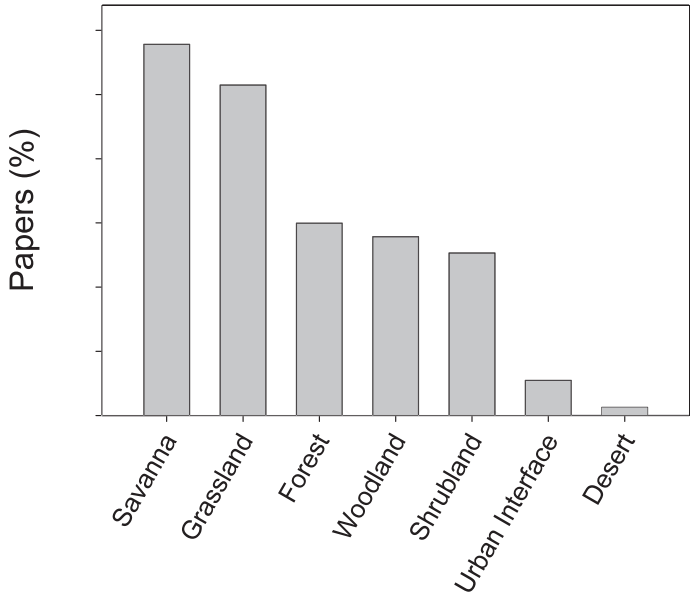
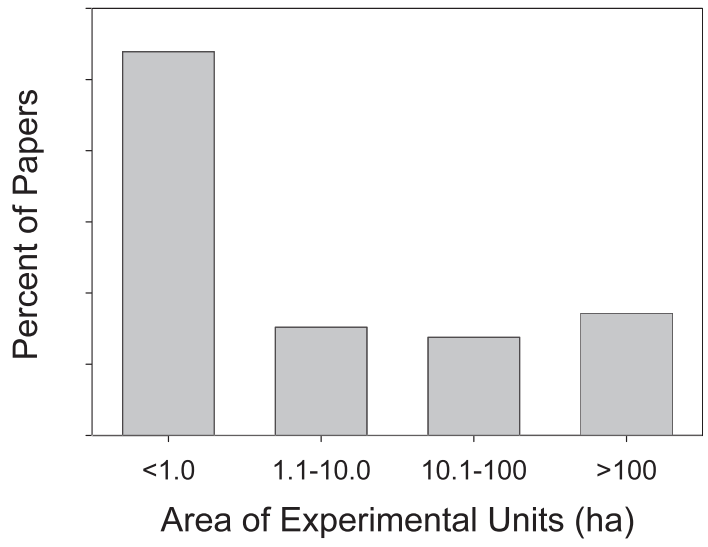


TABLE 4. Number of papers published reporting research on topic areas conducted within geographic regions of the United States based on a Web of Science search.

Topic area	Not specific	Eastern forests and grasslands	Great Plains	Intermountain West	Southwest Deserts	West Coast	Total
Plants	4	9	22	8	13	12	68
Soil	1	2	8	2	2	2	17
Water	1	0	0	0	0	0	1
Wildlife	2	1	3	3	1	2	12
Arthropods	0	0	8	0	1	0	9
Livestock	1	0	0	0	0	0	1
Fire Management	4	4	0	7	1	9	25
Fire Behavior	5	3	2	1	2	1	14
Smoke Management	1	0	0	1	0	1	3
Socio-economics	10	2	2	7	1	6	28
Air	0	1	2	2	2	3	10
History	1	2	0	3	0	1	7
Health	3	0	0	4	0	3	10
Other	3	0	2	0	1	3	9
Total	36	24	49	38	24	43	214

FIGURE 3. Number of rangeland fire papers published in each of four spatial scales. Papers were published in peer-reviewed journals and located in a Web of Science search.



grazing and only 26% and 19% of the studies specifically stated that they included ungrazed and grazed sites, respectively. Grazing was a part of the experimental design in only 13% of the studies. Because the vast majority of rangeland is grazed, the failure of research to address the interaction of fire and grazing severely limits applying the research to support NRCS purposes for prescribed burning.

EVALUATION OF FIRE EFFECTS ON ECOSYSTEM COMPONENTS

Prescribed fire is currently conducted on rangelands for many reasons, but a primary purpose is to reduce encroachment of invasive woody plants (see Fire and Plants section, Composition Changes subsection). Because fire can both positively and negatively influence ecosystem components, fire should be evaluated from the perspective of all ecosystem components. Therefore, we evaluate the literature available on plants, soil, water, wildlife, arthropods, livestock, fire management, fire behavior, smoke management, socio-economics, air quality, and fire history. Although we will evaluate the entire dataset when appropriate, on occasion we focus on data from specific rangeland regions of the United States to illustrate differences and similarities between regions.

Fire and Plants

The data set included 220 papers focused on plants and plant communities. Nearly 25% of the papers on plants evaluated community composition and structure following prescribed fire and wildfires. Ten percent of the papers evaluated biomass production, 6% discussed plant diversity, 5% considered mortality to individual plants, 5% focused on seed germination and establishment, and 2% considered plant (forage) quality. The remainder of the topics considered (plants as fuel, physiology, invasive species, harvesting, seed production) and were all below 2% of the papers evaluated and no conclusions could be drawn on these topics.

Biomass and Forage Quality. Plant productivity, plant nutrient content, plant diversity, and plant mortality responses to fire are highly variable (Blair 1997; Reich et al. 2001). Much of this variability depends on how productivity is defined. Forage or herbaceous production usually decreases for 1–2 yr after fire followed by a positive to neutral effect. An exception to this occurs in the tallgrass prairie where productivity can be enhanced from one spring fire following several years without fire (Blair 1997). Total aboveground biomass appears to be negatively associated with fire frequency although this depends on the ecosystem. Several studies indicate an increase in forage quality in mesic grasslands following fire (Hobbs et al. 1991; see wildlife and livestock sections), but these patterns are uncertain in more xeric regions. Grazing animals throughout all regions select recently burned areas for foraging sites over unburned sites, reflecting improved forage quality following fire. At a minimum, fire maintains openness of grassland landscapes, which sustains herbaceous biomass and forage quality/quantity. Because individual species and communities differ widely in response to fire, species diversity and plant mortality following fire cannot be generalized. Some evidence supports the NRCS purpose of using fire to enhance seed production, germination, and establishment, but these patterns are also species- and site-specific and cannot be generalized over all plants and ecosystems.

Compositional Changes. The majority of plant papers focused on community

composition. There were 68 papers in the data set that evaluated vegetation dynamics and plant responses to fire within the United States. We evaluated 81 papers to identify regional differences and similarities among the Great Plains, Intermountain West, and the Desert Southwest. Of the papers reviewed, 65% reported results of prescribed burns, 21% reported results of wildfires (Fig. 4), and most (> 75%) fire treatments were applied in spring and summer. Several studies recognized that season of fire mostly had a minimal or temporary effect (Engle and Bidwell 2001). A major concern from the database is the limited number of long-term studies (Fig. 5). Twenty-two percent of the studies extended past 10 yr, of which a majority substituted space for time (comparing plant succession across fires of different ages), with the majority of studies (64%) not extending beyond 3 yr.

Plant response to fire was highly variable both across and within regions and ecological sites (Table 5). A large portion of this variability can be attributed to the interaction of multiple variables, which include site characteristics, fuel characteristics, climate, community composition, time since fire, fire season, fire intensity, and postfire management. However, several patterns are evident in fire related plant responses across regions and ecological sites. *Perennial grasses* declined in abundance in the first postfire growing season in 76% of the studies, but usually recovered within the second or third year. Abundance of perennial grasses increased in only 11% of the studies in the first postfire growing season and in 5% during the second or third year following fire, but no studies reported long-term declines in perennial grasses. *Annual grasses* were usually more abundant in the first, second, and third years following fires compared to unburned stands. *Annual and perennial forbs* were inconsistent in their response the first year following fire, but they were more abundant in four out of six studies by the second or third year. Abundance of both resprouting and nonsprouting *shrubs* (biomass, cover, or volume) was lower during the first 10 yr following fire. However, density of sprouting shrubs usually equaled or exceeded that of unburned communities within 3 yr following fire suggesting little or no mortality. Full recovery of sprouting shrubs occurred within 3–20 yr; recovery took 25–35 yr for nonsprouting shrubs on relatively wet sites (e.g., mountain big sagebrush [*Artemisia tridentata* subsp. *vaseyana*] in the 300–400-mm precipitation zone) and greater than 45 yr on dry sites (e.g., Wyoming big sagebrush [*A. t.* subsp. *wyomingensis*] in the 200–300-mm precipitation zone).

The use of fire to increase cover, density, and biomass of herbaceous vegetation, particularly perennial grasses, is only weakly supported in

FIGURE 4. Number of rangeland fire studies addressing plant responses to prescribed fire and to wildfire in three rangeland regions of the United States.

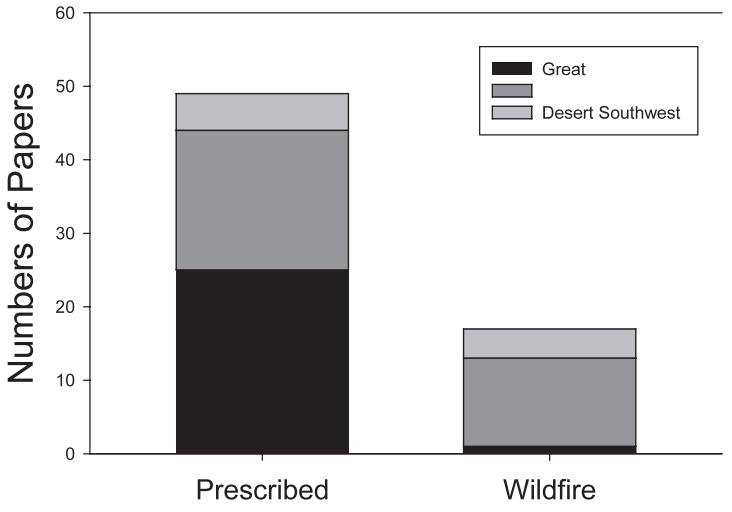


FIGURE 5. Time period over which individual research studies investigated plant response to fire on rangelands.

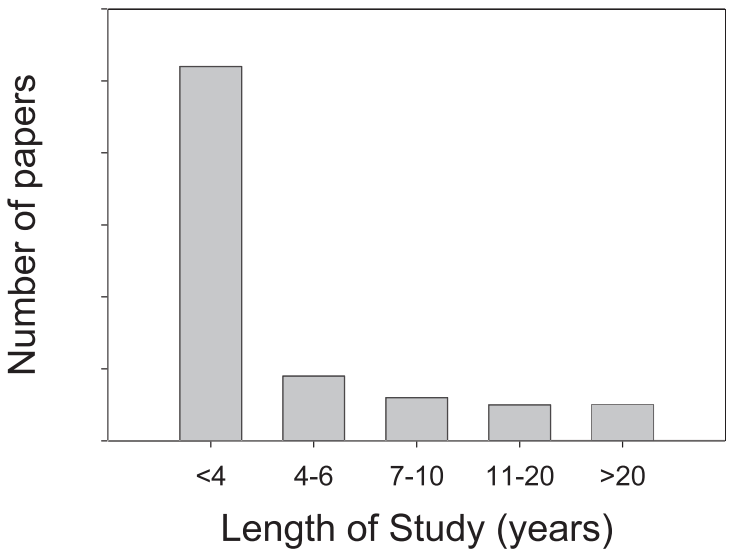


TABLE 5. Numbers of studies indicating negative (–), positive (+), and no change (=) in response of plant groupings (total herbs, perennial grasses, etc.) to fire across specific regions (Great Plains, Intermountain West, and Desert Southwest).

	1 yr postfire			2–3 yr postfire			≥ 4 yr postfire		
Plant grouping	(–)	(+)	(=)	(–)	(+)	(=)	(–)	(+)	(=)
Total herbs	7	4	4	2	3	3	0	1	1
Great Plains	7	4	3	2	2	3	0	1	1
Intermountain West	0	0	1	0	1	0	0	0	0
Desert Southwest	0	0	0	0	0	0	0	0	0
Perennial grasses	20	3	3	3	1	17	0	1	1
Great Plains	5	2	1	1	0	5	0	1	0
Intermountain West	10	1	2	1	1	7	0	0	1
Desert Southwest	5	0	0	1	0	5	0	0	0
Annual grasses	1	4	0	0	3	1	0	0	0
Great Plains	1	2	0	0	1	0	0	0	0
Intermountain West	0	2	0	0	1	0	0	0	0
Desert Southwest	0	0	0	0	1	0	0	0	0
Perennial forbs	2	3	0	0	3	1	0	0	0
Great Plains	1	0	0	0	1	0	0	0	0
Intermountain West	0	3	1	0	2	0	0	0	0
Desert Southwest	1	0	0	0	0	1	0	0	0
Annual forbs	2	2	1	0	1	1	0	0	0
Great Plains	1	0	0	0	1	1	0	0	0
Intermountain West	1	2	0	0	0	0	0	0	0
Desert Southwest	0	0	0	0	0	0	0	0	0
Shrubs, sprouting	4	0	0	1	0	0	4	1	1
Great Plains	0	0	0	0	0	0	0	0	0
Intermountain West	1	0	0	0	0	0	4	1	1
Desert Southwest	3	0	0	1	0	0	0	0	0
Shrubs, nonsprouting	9	0	0	5	0	0	0	0	0
Great Plains	0	0	0	0	0	0	0	0	0
Intermountain West	9	0	0	5	0	0	0	0	0
Desert Southwest	0	0	0	0	0	0	0	0	0
Trees, nonsprouting	4	0	1	2	0	0	1	0	0
Great Plains	0	0	0	0	0	0	0	0	0
Intermountain West	3	0	0	2	0	0	1	1	0
SW Deserts	0	0	0	0	0	0	0	0	0
Woody plants	3	0	0	4	1	0	4	1	2
Great Plains	3	0	0	4	1	0	4	1	2
Intermountain West	0	0	0	0	0	0	0	0	0
Desert Southwest	0	0	0	0	0	0	0	0	0

the research literature. However, possibly the strongest argument for the use of prescribed fire is to maintain or restore a desired successional community or pattern of communities in different stages of recovery following fire. Although herbaceous vegetation rarely increases the year of the fire, herbaceous dominance over woody plants is favored by shorter fire-return intervals. A number of studies across US rangelands reported that shrubs and trees increase and herbaceous vegetation decreases with long-term fire removal, so maintaining or increasing relative abundance of herbaceous vegetation requires periodic fire. The length of the fire-return interval can also determine the proportion of area occupied by grassland, shrub-steppe, and conifer woodland communities.

Fire can be used to change plant composition (e.g., the proportion of C_3:C_4 plants, herbaceous:woody plants, and forbs:grasses) and reduce excessive litter buildup resulting in an increase of light to basal tillers (an issue restricted to highly productive sites). The literature is mixed on one of the greatest concerns over the use of prescribed fire—the potential for increasing invasive species. Fire can act as a trigger to force a desirable stable-state that may be at risk of resilience loss across a threshold to an undesirable invasive plant state. Cheatgrass provides an excellent example of this dynamic in the Intermountain West. On the other hand, fire can be used to control invasive species through direct control or by focusing herbivory on a relatively small burned area within a landscape (Cummings et al. 2007).

Great Plains. Of the 36 papers reviewed from the Great Plains, almost all were prescribed fires at the sublandscape level (plots and stands) evaluating burns during the spring and summer. Several studies also evaluated the timing of burning in the spring (early, middle, and late) and reported a significant effect on vegetation response. However, a literature review on the effect of season of burn on herbaceous species in tallgrass prairie suggested that the data were not conclusive (Engle and Bidwell 2001). Only a limited number of studies reported the method of burning or the prefire and postfire conditions. Nearly two-thirds of the studies were less than 3 yr in duration with only 14% exceeding 10 yr.

Production and composition of herbaceous communities following fire were highly variable across studies. In the first growing season following fire, total herbaceous vegetation was less abundant in seven studies, more abundant in four studies, and did not differ from unburned in three studies. In one study burning increased photosynthesis and nitrogen uptake in perennial grasses in the first year, but biomass was less on burned plots than on unburned plots. In the second and third postfire growing seasons (most often the third year), herbaceous plant abundance generally increased to equal or exceed that of unburned plots. Late spring fires (May) often increased biomass of tallgrasses whereas early spring fires reduced biomass. Spring fires favored late flowering and C_4 plants, whereas summer fires favored C_3 and early flowering plants. Timing of burning also influenced the proportion of grasses and forbs. Fire appeared to have an extended negative effect on herbaceous biomass if drought followed in the first season postfire. However, one study reported only a very weak relationship between fire, weather, and plant response. Perennial grass biomass usually increased following fire in productive tallgrass sites where excessive accumulation of mulch occurs. Forb production was often reported to be greater on unburned plots.

Shrub abundance (biomass, cover, volume) consistently declined the first year following fire and was generally less than the controls 3 yr after fire. Density of resprouting shrub species recovered or exceeded preburn levels within 3 yr following fire. Most shrubs in the Great Plains are resprouting and fire return intervals of 2–5 yr may be required to maintain herbaceous dominance. Nonsprouting encroaching trees, primarily Ashe juniper (*Juniperus ashei*) and eastern redcedar (*Juniperus virginiana*), increase without fire and gain dominance after about 30 yr. The effects of these fires depend on grazing intensity, which constrains fuel load and fire intensity.

Intermountain West. Of a total of 36 studies reviewed, 32% investigated individual plant species and 73% emphasized plant community responses. The majority of studies evaluated summer burns (57%, many of which were wildfires), but more than half considered fall or spring burns (35% fall, 21% spring) and only a single study evaluated winter burning. Spring and fall burns were prescribed fires. The majority of studies were short-term (72%) and only 8% exceeded 10 yr. Very few studies reported the method of burning or prefire and postfire conditions.

Thirteen studies reported total perennial grass response in the first postfire growing season, of which 10 reported a decline in cover, biomass, or density; one an increase in cover; and two no change in cover. The majority of these studies (7 of 9) showed that perennial grass recovered to that of unburned plots within 2–3 yr, whereas one study showed a decline in cover and one study an increase in cover. Perennial forbs generally increased as did annual grasses in the first postfire growing season. Bluebunch wheatgrass (*Pseudoroegneria spicata*), Sandberg bluegrass (*Poa secunda*), and squirreltail (*Elymus elymoides*) were the most resistant to fire whereas Idaho fescue (*Festuca idahoensis* Elmer), Thurber's needlegrass (*Achnatherum thurberianum*), and rough fescue (*Festuca campestris*) consistently declined in the first year and either remained lower or recovered densities or cover equal to that of unburned plots. Broadleaved grasses and smaller bunches typically were more resistant to fire than fine-leaved grasses or large bunches.

Nonsprouting shrubs, primarily mountain big sagebrush and bitterbrush (*Purshia tridentate*, a weak resprouter), consistently decreased with fire and did not recover for 25–35 yr. However, recovery of Wyoming big sagebrush usually took longer with one study reporting only 5% sagebrush cover after 23 yr following fire. Cover of all shrubs (sprouting and nonsprouting) was reduced after fire. Few studies have evaluated sprouting shrubs (yellow and rubber rabbitbrush [*Chrysothamnus viscidiflorus* and *Ericameria nauseosa*], horsebrush [*Tetradymia* spp.]). However, limited work indicates biomass declines of these species in the first several years following fire with density typically recovering to preburn levels within 3 yr. In the Southwest, sprouting shrubs were the dominate vegetation 25 yr following fire.

Juniper (*Juniperus* spp.) and piñon pine (*Pinus* spp.) densities are reduced following fire, but large trees are more difficult to kill than small trees. Juniper cover of individual trees

> "A number of studies across US rangelands reported that shrubs and trees increase and herbaceous vegetation decreases with long-term fire removal…"

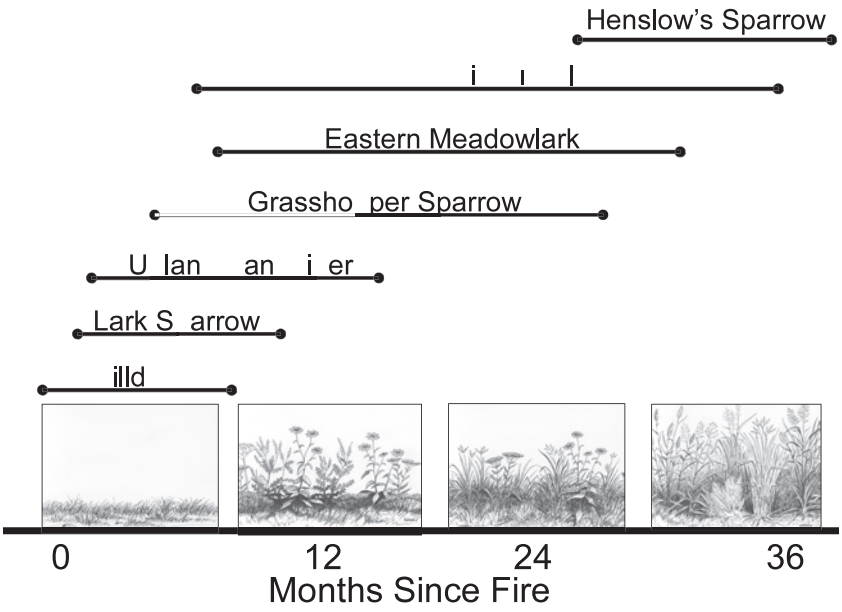


FIGURE 6. Response of grassland birds to time since focal disturbance by fire and grazing at the Tallgrass Prairie Preserve, Oklahoma, 2001–2003. Some birds native to the area require recently burned patches that are heavily grazed whereas others require habitats that are undisturbed for several years (Fuhlendorf et al. 2006). This research emphasizes that 1) the response of grassland birds to fire is highly dependent upon the interaction of fire and grazing and 2) fire management should not be considered in isolation from other environmental factors, including grazing. Figure courtesy of Jay Kerby and Gary Kerby.

increases slowly for the first 45 yr followed by rapid increase during the next 46–71 yr. Closed canopies can develop within 80–120 yr (Johnson and Miller 2006). Fire in woodlands is typically followed by an increase in perennial grasses and a reduction of woody plants. Sagebrush (*Artemesia* spp.) reached preburn levels in 36 yr and then often declined if piñon pine and/or juniper became established on the site. Understory cover declined to 5% of the adjacent grassland by 100 yr following fire as piñon and juniper woodlands developed (Barney and Frischknecht 1974; Wangler and Minnich 1996). However, understory composition following fire is highly dependent on the composition and abundance of the understory prior to the burn.

Fire and Wildlife

Of the 40 papers we evaluated concerning the effect of fire on rangeland wildlife, only 12 papers addressed US rangeland wildlife. These 12 papers focus on avifauna and small mammals, reflecting the large influence exerted by fire on habitat structure, to which these vertebrate assemblages are especially sensitive. It is interesting to note that measurements in most wildlife studies, including those we sampled, focus on wildlife population response and relatively few (only 2 of the 12 US studies) measured vegetation attributes (e.g., horizontal and vertical structure). Ten of the 12 studies were published since 2000, which indicates a recent upswing in research interest in wildlife response to fire. However, only 2 of the 12 studies included private land. The 12 studies were spread more or less evenly across geographic regions and vegetation types. As might be expected from a small number of studies, the studies addressed only a small number of questions related to the fire regime and the grazing environment. For example, only one of the studies (Fuhlendorf et al. 2006) addressed the ecological interaction of fire and grazing. From the Fuhlendorf et al. (2006) study and related research, we know that the ecological interaction of fire and grazing strongly influences habitat selection and habitat value for virtually all rangeland wildlife, including birds (Churchwell et al. 2008; Coppedge et al. 2008) and large ungulates (Hobbs and Spowart 1984; Coppedge and Shaw 1998; Biondini et al. 1999; Van Dyke and Darragh 2007). Large herbivores are attracted to nutritious regrowth of herbaceous vegetation, sometimes emerging outside the growing season (Coppedge et al. 1998; Biondini et al. 1999) on recently burned areas (Hobbs and Spowart 1984; Hobbs et al. 1991; Turner et al. 1994). In contrast, many, but not all, rangeland small mammal and bird species are more suited to areas not recently burned and grazed because these areas provide vegetative cover required for concealment or nesting. However, this influence can be mediated by drought (see Meek et al. 2008) and other factors.

The context in which prescribed burning is applied on rangeland marks the effect on wildlife species in question. Wildlife species in a given area have variable habitat requirements, so positive response by one species will likely cause other species to decline (Fig. 6). However, because rangeland and rangeland wildlife evolved with periodic fire and because periodic fire is required to maintain habitat suitable for wildlife species native to a particular rangeland region, fire is essential for maintaining rangeland wildlife populations.

Unnaturally long fire-return intervals often lead to tree encroachment and other changes that reduce habitat suitability for native wildlife species that are habitat specialists (Coppedge et al. 2001; Reinkensmeyer et al. 2007), some of which are species of conservation concern. In contrast, fire-return intervals greater than those with which an ecosystem evolved can have correspondingly deleterious effects on habitat and populations of habitat specialists (Robbins et al. 2002; Pedersen et al. 2003; Fuhlendorf et al. 2006; Rowland et al. 2006).

Fire and Water

We reviewed 25 papers that evaluated fire effects on various hydrologic processes in rangeland (Table 6). Hydrologic variables evaluated were water repellency (six papers), water quality (two papers), hydraulic conductivity or infiltration (six papers), and erosion/runoff (five papers). The majority of studies were conducted for 3 yr or less: 1 yr (52%), 2 yr (20%), and 3 yr (12%). Three studies (12%) were conducted for 5–6 yr, and one study was conducted for 9 yr. Variables that influenced the effects of fire on hydrology were aspect, fire severity, and microsites (coppice dunes formed beneath shrubs and trees vs. interspace). The largest decrease in infiltration rate and increase in erosion following fire occurred in coppice dunes beneath shrubs and trees. Fire had little effect on these two variables in shrub or tree interspaces. Water repellency usually occurred on both burned and unburned sites but usually increased, particularly beneath shrub and tree canopies, following fire. Hydrophobicity was reported to decline within several months to near preburn levels following wetting. Soil erosion on cooler, wetter sites in sagebrush-steppe communities (e.g., north aspects, or sites occupied by Idaho fescue compared to bluebunch wheatgrass) were less affected by fire than drier, warmer sites. One study reported rill erosion as the primary source of sediment and several studies reported rills readily formed in the coppice dunes. In general, these studies suggest that immediate effects of fire are largely negative on watersheds, but that the effects are short-lived.

Fire and Arthropods

Eighteen studies and one extensive literature review were evaluated for the effects of fire on arthropods. The majority of studies evaluated the response of grasshoppers (six studies) or arthropods in general (n = 6) to fire. Other species studied were ants (three studies), beetles (one study), and ticks (one study). In a literature review, Swengel (2001) reported few studies were conducted at the species level. The response of insects to fire was highly variable. Short-term and long-term response of insects to fire was influenced by intensity, complexity or patchiness of the burn, species requirements, and plant recovery. Thirteen of the 17 studies were conducted for 3 yr or less, three studies ranged from 4 yr to 9 yr, and one study extended for 25 yr. Insect abundance was usually lower (with the exception of grasshoppers) immediately following fire (up to 1–2 mo). In a 7-yr study across 21 different Great Plains sites, Panzer (2002) reported 93% of the species were consistent in their response to fire over the period of the study. Immediately following fire, 26% of arthropod species increased and 40% decreased. Of those that declined nearly two-thirds recovered within 2 yr. Insect orders *Homoptera* and *Hemioptera* appear to generally be more sensitive to fire whereas *Orthoptera* was little affected by fire. Fire effects on grasshopper populations generally showed limited response, but a shift in species composition frequently occurred.

Fire History

Obtaining a clear picture of the complex spatial and temporal patterns of historic fire regimes across the western United States before Euro-American settlement is unlikely. This can be attributed to limited sources of material (e.g., large charred wood or fire scars) available for reconstructing pre-historic fire regimes on most rangelands and the vast variation in fuel composition and structure,

TABLE 6. Effects of fire on several hydrologic variables compared to unburned plots and time periods required for recovery to near preburn levels.

Variable	Decreased	Increased	No change	Recovery
Soil repellency	1	5	0	2.5–3 mo
Infiltration	6	0	4	2 yr
Runoff	0	6	1	2 yr, 4–5 yr
Sediment loads	0	9	1	1 yr
Water quality	2	0	0	3–5 yr

In many cases, when woody plants reach a substantial size and/or density, fires will either be ineffective or require greater expertise to conduct them. Using extreme fires for restoration is an emerging topic in conservation of rangelands. An active prescribed burning program could help minimize the risk associated with extreme fires. (Photo: John Weir)

landscape heterogeneity, ignition from aboriginal and lightning sources, weather, and topography across this region. We reviewed 24 papers that attempted to describe pre-historic fire histories across the western United States. We tried to capture all of the papers that collected quantitative data to reconstruct pre-historic fire regimes related to rangelands. We also included several studies in ponderosa pine in addition to woodlands that evaluated the timing of reduced fire occurrence and livestock grazing. Twelve of the papers were based on fire scar data, two used charcoal or ash samples, and three used the presence of old-growth piñon or juniper trees. Of the 12 papers using fire scar data, samples were cross-dated in seven studies. Cross-dating is a procedure that verifies the exact year of the fire, important for calculating fire-return intervals and determining the extent of individual fires across larger areas.

Duration of the fire record based on fire scars ranged from 250 yr to 500 yr before present (BP). Charcoal studies ranged from 5 500 yr to 6 000 yr BP. Pre–Euro-American settlement fire regimes reconstructed from cross-dated fire scars or large charred wood across western rangelands are few and primarily restricted to the intermountain region. Fire histories based on fire-scarred trees are also spatially limited to the rangeland–forest ecotones in the Intermountain West, which often occur as mosaics of conifers and sagebrush-steppe grasslands. Fire-scar samples are usually collected from fire-resistant trees

(e.g., ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) and occasionally less fire-resistant trees (e.g., Utah juniper [*Juniperus osteosperma*], piñon pine). Several of these studies also evaluated tree age structure in adjacent forest and shrub-steppe patches.

Pre-historic (pre-1900) mean fire-return intervals reported along ponderosa or Douglas-fir–mountain big sagebrush-steppe ecotones varied from less than 10 yr (three studies) to 10–30 yr (six studies). Studies reporting longer fire-return intervals were associated with low sagebrush (*Artemisia arbuscula*; 90 yr to 150 yr) and were based on tree age structure and charred logs and stumps of juniper. The relatively short fire-return intervals (< 30 yr) would have supported grass-dominated communities along the forest ecotones. Extrapolation of these fire-return intervals away from range–forest ecotones is probably speculative and likely becomes longer in more arid ecological sites, especially those occupied by Wyoming big sagebrush. Macroscopic charcoal data collected in central Nevada suggested that fire-return intervals in the drier Wyoming big sagebrush cover type over the past several thousand years were up to a century, with fire intervals varying with climatic fluctuations.

Several consistent patterns regarding fire-return intervals emerge from these papers. First of all, there is consistent evidence that most rangelands in the United States have experienced a dramatic increase in fire-return intervals over the past 100–200 yr. Six of the studies reported sharp declines in fire occurrences that coincide with the introduction of livestock. Piñon–juniper woodlands that have persisted for the past several or more centuries did not show evidence of high-frequency, low-intensity surface fires. Five of the studies reported probability of sites being occupied by old-growth trees to be associated with rocky surfaces and limited surface fuels but none as fire refugia. Three studies reported the probability of large fires increases in years preceded by wetter than average years. At a longer time scale, Mehringer (1987) and Mensing et al. (2006) reported a correlation of increased fires during periods of wetter than average conditions.

Quantitative measures of pre-historic fire return intervals in the tallgrass prairie are not available for the Great Plains. The assumption that prehistoric fire regimes in the tallgrass prairie were characterized by frequent low-intensity fires is primarily based on 1) observations from early explorers, trappers, and settlers, and 2) research showing that in the absence of fire these grasslands shift rapidly from prairie to woody species (Bragg and Hulbert 1976). Several authors have estimated mean fire-return intervals of 3–5 yr (Wright and Bailey 1982; Knapp and Seastedt 1996). However, little is known about the dynamics of native grass and woody species prior to Euro-American settlement. It is also likely that the influence of bison on fuel loads affected fire-return intervals across the Great Plains.

Fire and Soils

The vast majority (45 of 51; 88%) of the papers we evaluated on fire effects on rangeland soils were published in ecological or soil science journals rather than *Journal of Range Management*, *Rangeland Ecology and Management*, or applied ecology journals. Therefore, the overall emphasis within the research base leans toward ecological understanding rather than to explicitly answering management questions. Twenty-eight papers (55%) reported effects of fire on soil chemistry (pH, nutrients), and 17 (33%) reported on the effects of fire on one or more variables (infiltration, soil water content, water repellency, erosion) related directly or indirectly to the water cycle.

The literature on rangeland soils, including the effects of fire on rangeland soils, is quite voluminous. For example, one of the sampled papers is a recent analysis of the literature on water repellency. In it, Debano (2000) employed a bibliography of over 700 published papers reporting on either various aspects of water repellency (500 papers) or published papers (200) that contribute information directly related to understanding the basic processes that underlie soil water repellency. Water repellency, a global rangeland issue reported for numerous vegetation types following fire, also occurs in soils other than rangeland.

The scope of these studies further limits the inferential base for applying the results to

“…there is consistent evidence that most rangelands in the United States have experienced a dramatic increase in fire-return intervals over the past 100–200 yr.”

Bison at the Tallgrass Prairie Preserve in Oklahoma grazing on a recently burned patch. (Photo: Steve Winter)

management of US rangelands. Although physical processes are not place-bound, only 16 of the 51 studies reported research from US rangelands. Fortunately, these were distributed more-or-less evenly across the United States (eight in the Great Plains or central prairies) and across vegetation types (grasslands, shrublands, etc.). However, small plots (0.0003–1 ha) were the general rule and studies often reported effects from a single fire (22 papers), and only 10 of the 51 studies encompassed time periods of 10 yr or more. An encouraging sign is a recent increase in published studies enhancing basic understanding of soil response to fire; the majority of papers published since 1998 (35; 69%) focus on this.

The influence of fire on soil depends largely on the prefire and postfire environment, interaction with other factors including grazing and invasive species, and the evolutionary history of the ecosystem with regard to fire frequency and grazing. However, it is notable that on US rangeland that are often characterized as lacking a long evolutionary history of frequent fire most fire research is based on observations following wildfires rather than controlled studies with prescribed fires. For example, portions of the Great Basin shrub-steppe have had substantial increases in fire-return interval and burn area over the past century (Miller et al. 2011). In contrast, prescribed burning and the ecological role of fire are the context of studies on rangelands characterized by a long evolutionary history of frequent fire, specifically the Great Plains.

Soil organic matter, resistant to change when rangeland fire is wind-driven and fueled by fine fuel, has long been a subject of interest to rangeland fire researchers (e.g., Reynolds and Bohning 1956; Owensby and Wyrill 1973). Recent research has increasingly reported the influence of fire on soil organic carbon (and CO_2 ecosystem flux) and carbon sequestration, which is tied to atmospheric properties related to global climate change. In the single study

located in the United States that appeared in our sample (Ansley et al. 2006a), carbon storage in soil increased with fire, likely the result of a shift in species composition. Recent research in the Intermountain shrub-steppe suggests plant invasions (i.e., *Bromus tectorum*) can reduce soil carbon (Bradley et al. 2006; Prater and DeLucia 2006), but this did not occur in a similar shrub community when perennial native grasses dominated postfire (Davies et al. 2007). Burning changed soil carbon in a semiarid Great Plains rangeland, but the magnitude of change was inconsequential partly because of a relatively low CO_2 flux (MacNeil et al. 2008). In subhumid Great Plains rangeland where CO_2 flux is markedly greater, soil carbon flux increases with periodic burning over ungrazed rangeland because burning removes accumulated litter that creates temperature and light-limiting conditions for plant growth, but annual burning will reduce both soil organic matter and nitrogen mineralization (Ojima et al. 1994; Blair 1997). Annual burning over perhaps 20–100 yr may increase the fraction of passive soil organic matter at the expense of more active fractions, which might ultimately reduce total soil organic carbon (Ojima et al. 1994). Nevertheless, most ecosystem carbon loss in rangeland results from combustion of aboveground organic material (i.e., fuel), with the time to reach prefire levels dependent on primary productivity (MacNeil et al. 2008). Soil carbon response to the ecological fire–grazing interaction has not been investigated, but because nitrogen and carbon are coupled in the organic matter pool, soil carbon might show similar increases following fire–grazing disturbances to soil nitrogen (Anderson et al. 2006).

Nitrogen in aboveground biomass is volatilized in fire and varies with environmental conditions and fire characteristics in that drier fuels and soils and hotter fires result in more intense combustion and more nitrogen volatilization (DeBano et al. 1979). Because most prescribed burning objectives call for conditions that consume most aboveground herbaceous fuel, it is often assumed that fire depletes ecosystem nitrogen. Indeed, postburn soil inorganic nitrogen (NO_3 and NH_4) is often less, but greater herbaceous aboveground annual production and vegetation cover at some point after burning suggests plant-available nitrogen increases following burning.

Stocker calves grazing on areas recently burned in the summer and providing new growth forage in the fall at the Stillwater Research Range. Unburned patch is shown in the background. (Photo: Sam Fuhlendorf)

> “…documentation of fire behavior on rangelands does not provide suitable guidance to address the NRCS’s purposes of prescribed burning.”

Research in subhumid rangeland (Blair 1997) and semiarid rangeland (Davies et al. 2007) indicates that burning increases nitrogen mineralization and enhances other mechanisms that result in increased nitrogen. Therefore, burning indirectly enhances plant capability to utilize nitrogen. Annual burning of subhumid rangeland over a period of 20–100 yr has been predicted to reduce mineralizable nitrogen similar to the effect on soil organic carbon (Ojima et al. 1994). Because grazing reduces the amount of nitrogen available for volatilization by fire and because nitrogen loss is proportional to biomass available for combustion, grazing weakens the effects of fire on soil nitrogen (Hobbs et al. 1991). This likely explains why nitrogen fertility was not diminished with annual burning coupled with long-term moderate grazing (Owensby and Anderson 1967). Moreover, this mediating effect of grazing is subject to the effect of scale and preferential grazing of patches (McNaughton 1984; Hobbs et al. 1991). When fire and grazing interact spatially (i.e., the fire–grazing ecological interaction) in a subhumid rangeland, plant-available nitrogen increases in recently burned, focally grazed patches (Anderson et al. 2006), but unburned patches with minimal grazing pressure have low levels of available nitrogen. No published research on the effects of the fire–grazing interaction on soil nitrogen is available for other rangelands.

Some fire prescriptions, wildfire, and fuel situations in rangeland can result in extreme soil heating, which can markedly change soil chemical and physical properties. Brush piles and thinning slash, in particular, create intense heat that can change biological, chemical, and physical properties of soil and can induce undesirable vegetation change including plant invasions (Neary et al. 1999; Haskins and Gehring 2004). Although the mechanisms and impact of soil heating are known, other than postfire restoration (for example, see Korb et al. 2004), we found no studies that fashion rangeland fire prescriptions and vegetation management guidelines to reduce the impact of soil heating with burning brush piles and slash.

Fire and Livestock

Among the 476 papers evaluated pertaining to fire on rangelands, over 25% of the studies referred to grazing by either livestock or wildlife. However, only five papers evaluated the influence of fire on livestock performance, of which only one paper was from the United States. Recognizing the potential for bias within our initial literature review, we broadened our search criteria to include articles with “fire or burn” and “livestock, cattle, sheep, or horses” in the subject. Even with this less restrictive search, only three additional papers focused on fire and livestock and they revealed a span of 26 yr between studies. Due to the limited number and geographic scope of the studies, few detailed inferences can be made. Early research from the Great Plains region focused on livestock performance and indicated that midspring and late spring burning increased steer weight gains, particularly early in the growing season (Anderson et al. 1970; Smith and Owensby 1976). Burning portions of unimproved Florida rangeland annually improved calf production as much as 75% over unburned pastures (Kirk et al. 1976). The increased benefit from burning alone was comparable to supplemental feeding, especially when considering the increased cost of supplemental feed. Increased livestock performance is mostly attributed to increased forage quality from increased plant crude protein and decreased fiber content following fire (Grelen and Epps 1967; Allen et al. 1976; Kirk et al. 1976). Within the fire–grazing interaction, cattle are attracted to recently burned locations, which provide higher-quality forage than the surrounding matrix of unburned vegetation, and devote 75% of their grazing time within recently burned areas (Fuhlendorf and Engle 2004). The disturbance created by the interaction of cattle grazing and fire mimics historical grazing behavior of large ungulates and creates a structurally diverse landscape.

Prescribed Fire and Air Quality/Smoke Management

Rangeland fire generates a wide variety of by-products that fall into two broad categories, gasses and particulates. Smoke, the visible product of partially combusted fuel material, contains an array of organic and inorganic airborne particulates. Airborne particles can be a nuisance, reducing visibility hundreds of kilometers downwind from emission sources (Ferguson et al. 2003; McKenzie et al. 2006) and degrading air quality (Martin et al. 1977).

Prescribed fire produces 15–25% of airborne particulates and 7–8% of hydrocarbons emitted to the atmosphere annually (Martin et al. 1977). However, environmental conditions, fuel characteristics, and characteristics of the fire itself influence the amount of noncombusted material produced. Relatively low-intensity fires, where more complete combustion is expected, were modeled to produce about 50% less smoke than higher-intensity fires with higher rate of spread and less complete combustion (Glitzenstein et al. 2006). Fuel consumption and fire intensity clearly influence emissions from rangeland fire.

Particle size influences the period of suspension in the atmosphere. Relatively large particles, between 0.07 µm and 1.0 µm diameter, may take days to settle out, whereas small particles less than 0.07 µm do not settle under natural conditions (Martin et al. 1977). Larger particles do not remain suspended for long time periods, yet can be problematic for individuals with asthma or other chronic respiratory conditions (Dockery et al. 1993). Although smoke management is important as it relates to air quality, our review of the literature revealed that only six papers addressed smoke management on rangeland and thus conclusive evidence is limited. Further investigation is needed to provide a complete understanding of how prescribed fire influences air quality.

Fire Behavior and Fire Management

In addition to contributing to an understanding of the influence of fuels and environmental conditions on fire behavior, many of the fire behavior studies that we reviewed more directly addressed questions related to other sections of this report. Only a few studies addressed plant responses (e.g., tree mortality; Kupfer and Miller 2005) as a function of fire behavior, and fire behavior was reported in several studies as one of several aspects of environmental conditions under which the study was conducted (e.g., Engle and Weir 2000; Ansley et al. 2006a). Measuring a correlate (e.g., char height on trees; Fule and Lauglin 2007) of a primary fire behavior characteristic (e.g., fireline intensity) was common (8 of 11 US papers).

We examined our sample of published papers to determine the extent to which they addressed NRCS's relevant objectives of prescribed burning (i.e., prepare sites for harvesting, planting, or seeding; reduce wildfire hazards; remove slash and debris). Of the 41 studies sampled on fire behavior, only 11 were located in the United States (and three were in southeastern forests), so specific application to US rangelands is minimal for at least three-quarters of the studies. Of the 11 US studies, two studies (Sparks et al. 2002; Glizenstein et al. 2006) related to wildfire hazard reduction and slash removal in southeastern US forests, and no studies were related to fuels management on rangeland. One study (Gilless and Fried 1999) examined a computer model for its utility in planning fire suppression. Based on our literature search, it appears documentation of fire behavior on rangelands does not provide suitable guidance to address the NRCS's purposes of prescribed burning. For example, mortality or scorch height of invasive woody plants is highly dependent on fire intensity, which is rarely measured in rangeland studies.

The refereed literature on fire management is insufficient to adequately evaluate the NRCS's purposes of prescribed burning (e.g., what types of management will promote different purposes). However, a significant quantity of literature on fire behavior and fire management, addressing both prescribed burning and fire danger related to rangeland is found in federal government documents, especially those published from the US Forest Service Fire Sciences Laboratory in Missoula,

Backfire of a prescribed burn at the Tallgrass Prairie Preserve of Oklahoma. (Photo: Terry Bidwell)

TABLE 7. Assessment of the 10 purposes within the Natural Resources Conservation Service (NRCS) prescribed burning conservation practice standard relative to the supporting experimental evidence. Observations on the evidence provided by the peer-reviewed scientific literature supporting NRCS purposes for prescribed burning.

Purpose	Aspects that agree with the purpose	Aspects that suggest limited or no support for purposes	Further needs and considerations
Control undesirable vegetation	Fire can be effective in reducing the stature of resprouting, fire-adapted shrubs and trees, some invasive herbaceous plants, and the encroachment of fire-sensitive shrubs and trees.	Most effects are too short-lived to support the purpose over meaningful management time spans. Most species are reduced only in stature and without mortality. Many species are not negatively affected by fire, and some species increase after fire.	Generalizations concerning season, frequency. and intensity of fire are mostly unsupported.
Prepare sites for harvesting, planting and seeding			No evidence in our database—additional searches[1] generated few additional papers with controlled comparisons.
Control plant diseases	One study from a comprehensive search[1] suggested that fire can reduce the density of host species (oak), but the authors recommended more study to validate their results.		No evidence in our database—additional searches[1] generated few additional papers with controlled comparisons.
Reduce wildfire hazards	Models and observational studies suggest that fire-induced vegetation changes can alter subsequent fire regimes and reduce fine fuel loads to decrease the likelihood of high-intensity, stand-replacing, destructive wildfire.	Prescribed fire that reduces woody plants and maintains grassland productivity can increase the likelihood of fire.	No evidence in our database—additional searches[1] generated few additional papers with controlled comparisons.
Improve wildlife habitat	Fire can maintain and restore habitat for native wildlife species in some situations.	Any action that improves habitat for one species likely degrades habitat for another.	Studies limited mostly to birds and small mammals. Time since fire, the key element, has been minimally studied.
Improve plant production quantity and/or quality	Several studies indicate increased plant production and forage quality but these are mostly restricted to the Great Plains—several additional studies indicate animal preference for burned sites. Most studies show an initial decrease in quantity and then recovery, with limited studies showing an actual increase over time. However, majority of studies do not evaluate response beyond 5 yr.		Minimal information on grazing animal response following fire. Therefore, if the purpose is intended to benefit livestock production through increased forage production or improved forage quality, support is limited. The use of terminology such as "plant quality" is overly broad and could suggest wildlife habitat is improved following fire, but this is not supported well in the research literature.
Remove slash and debris	Fire can be used in Southeastern pine forest to remove slash and maintain savanna and to remove brush piles and brush windrows.		No evidence in our database—additional searches[1] generated few additional papers with controlled comparisons. Because fire effectiveness varies, more study is needed on fire effects and fire management after brush treatments to restore rangeland.
Enhance seed production	Fire can increase seed production of both native species and exotic invasive plants.		Seed production is rarely measured in fire research. This purpose is irrelevant to those rangeland plants that reproduce vegetatively.
Facilitate distribution of grazing and browsing animals	Recently burned areas generally attract grazers because burning increases herbivore access to current year's forage growth.		The fire–grazing interaction is an appropriate tool that employs attraction of large herbivores to recently burned areas, but research is limited to experimental studies in Oklahoma and observational studies of wildlife.
Restore and maintain ecological sites	Fire regimes that mimic evolutionary conditions of the rangeland in question will maintain ecological sites and therefore maintain grassland, savanna, and shrubland ecosystem structure.	Reintroducing fire will not always restore ecological sites. Prescribed burning that creates fire regimes that do not mimic historic fire regimes can induce site degradation by altering biotic and abiotic characteristics (e.g., hydrophobic soil).	Most ecological sites lack a complete fire regime description (especially where fire intervals are long).

[1]Additional literature searches used a topic search in Web of Science for *fire* AND *rangeland* AND *Reduce wildfire hazards*.

Montana. The early science on fire behavior that culminated in Rothermel's (1983) seminal user-friendly fire behavior prediction model was applied to rangeland, and many rangeland fire managers have used this version of the model that does not require a computer. More recently, a suite of dynamic computer models have greatly expanded the management value of the Rothermel model to applications in variable terrain and varied fuels and fuel loads. Coupling these models with sophisticated fuel models informed by state-of-the-art fire–weather observing stations and near real-time remote sensing of fuels has greatly enhanced fire management on privately owned rangeland (Carlson et al. 2002; OK-Fire 2009).

Fire and Human Dimensions

From all of the papers evaluated, 36 addressed a wide variety of human aspects related to rangeland fire ranging from education and public perception of fire to health and policy issues. Although all regions of the United States were covered, over 80% of the studies focused on the West Coast or Intermountain regions. Human dimensions on rangelands have recently gained attention and are reflected in 75% of the studies pertaining to fire dated 2000 or later. Because of the recent interest in human dimensions, CEAP devoted an entire section to socioeconomics, so we restricted our summarizations to limit duplicate information.

DISCUSSION OF FINDINGS

Most research evaluated here was not developed with the intent of providing specific recommendation for management of rangeland landscapes. Moreover, constructing research based on NRCS purposes for prescribed burning is limited by spatio-temporal scale of the research; limited description of conditions prefire, postfire, and during the fire; failure to account for interaction with other disturbance processes occurring on rangelands (e.g., grazing, drought); simplifying a complex fire regime to a single treatment event; and the lack of a focus on fire effects that are highly dependent on time since fire. Even with this paucity of research evidence, our evaluation demonstrates that several of the NRCS purposes for prescribed burning can be justified but with many caveats (Table 7). Specifically, management of woody plant invasion is supported by a fairly consistent response in which prescribed fire limited invasion. It is less clear if fire can reverse woody plant invasions when thresholds have been crossed. The purpose of using prescribed burning for short-term control of undesirable plants is justifiable based on research that shows some herbaceous species respond negatively to fire in the first growing season following fire, especially when combined with focal grazing. However, most herbaceous species recover within 2–3 yr postfire regardless of season of the burn. Contrary to this purpose, some regionally important herbaceous species in each rangeland region of North America respond negatively to fire 1 yr, 2 yr, and 3 yr postfire. In general, few negative effects and more neutral and positive effects have been demonstrated on target herbaceous species in response to fire. With the exceptions of the expansion of invasive annual grasses in the Intermountain West following fire (Miller et al. 2011), these conclusions were surprisingly consistent across the Great Plains, Intermountain West, and Desert Southwest. A few studies report increased productivity and forage quality the year of the fire, but because these are mostly from mesic grasslands, this NRCS purpose for prescribed burning is not broadly supported by research evidence.

It is widely recognized that the importance of all processes on ecosystem structure and function are highly dependent on the scale of observation. In fact, it has been suggested that studies should be conducted at multiple scales and the interpretations of research should recognize the limited ability to translate conclusions across scales. Several rangeland studies have demonstrated that vegetation dynamics (Fuhlendorf and Smeins 1996, 1999; Briske et al. 2003) and wildlife populations (Fuhlendorf et al. 2002) are highly dependent on spatial and temporal scale and that important processes at one scale are not necessarily transferable to other scales. Our analysis suggests that the vast majority of the data available on fire is either conducted at scales too small to be relevant to management (< 1 ha) or, in some regions, based on wildfire. Most studies were limited to short-term responses (< 3 yr) and often a single application of a fire, so they have minimal application to the long-term goal of restoring fire to

> "…most herbaceous species recover within 2–3 yr postfire regardless of season of the burn."

“Experimental fire research rarely treats fire as a *regime* in which fire recurs and response to fire is dynamic and variable with fire intensity, fire interval, and other fire variables.”

rangeland ecosystems and strategic ecosystem management.

Conducted largely as short-term studies on small plots, much of the research on prescribed burning is unable to describe complex patterns in space and time that may be associated with interactions with other disturbances, such as grazing and drought. Because these studies are but a step removed from highly controlled greenhouse studies with limited application in the real world, much of this research fits solidly within a “So what?” category when evaluated for specific management application. For example, a study conducted on vegetation response to burning that is conducted by mere coincidence in a drought and on small plots that were not grazed (e.g., Engle and Bultsma 1984) cannot be used to support prescribed burning relative to vegetation responses across spatially variable, grazed pastures in nondrought periods.

Long-term research at management-relevant scales that embrace interactive responses and complex patterns is insufficient to provide the NRCS with data necessary for constructing the relevant purposes for prescribed burning. The synergistic effects of fire and grazing on large landscapes have largely been uncoupled within rangeland research and conservation, even though most are aware that grazing by native and domestic herbivores is a dominant feature on all rangelands. An example of the uncoupling of fire and grazing is the tendency to recommend removal of grazing following fire, which does not seem to be supported by research. Grazing was part of the experimental design of fire studies in a mere13.3% of the studies and most of the studies inadequately presented the methods or results to provide effective conclusions regarding the presence or absence of grazing. Fire and grazing operated historically as an interactive disturbance process in which neither was independent of the other. When allowed to interact in space and time, fire and grazing create a shifting mosaic pattern that cannot be predicted from short-term, small-scale studies (Fuhlendorf and Engle 2001, 2004). Understanding the effects of a fire on grassland soils is highly dependent upon grazing (Hobbs et al. 1991). Moreover, nonequilibrial dynamics and resilience theory predict that episodic events, such as drought, can interact with other processes, such as fire and grazing, to promote changes that may not be predictable from understanding these events independently. Finally, most research on burning compares two treatments, *fire* and *no fire* (or burned and unburned), and the fire treatment is a single event rather than reoccurring fire couched within a complex fire regime.

Much of the research also lacks relevance to prescribed burning as a conservation practice because it fails to account for the potential for fire effects to be highly dynamic and variable with time since the previous fire. Only 4% of the papers included *time since fire* as an important variable when describing the magnitude and persistence of fire effects. This synthesis indicates that vegetation response 1–3 yr since the previous fire differs considerably from vegetation response in which the most recent fire was 4 yr ago or longer. Moreover, recovery rate varies greatly among response variables. Recovery of soil and water variables can be as little as several months to as much as decades depending on factors such as soil structure, vegetation condition at the time of the fire, and fire intensity. Fire frequency also compounds temporal response to fire, but it was a primary focus in only 12.5% of the studies. This further emphasizes that the research largely fails to assess dynamic fire regimes and the long-term dynamics of fire-dependent ecosystems.

Experimental fire research rarely treats fire as a *regime* in which fire recurs and response to fire is dynamic and variable with fire intensity, fire interval, and other fire variables. Research and management often approach fire as a single discrete event, so the impact of fire is highly dependent upon the conditions at the initiation of fire, conditions following fire, fire season, fire intensity, and time since fire. Research on fire regime rather than on discrete fire events would be more comparable to the study of grazing systems or constant stocking rates rather than the study of a single plant defoliation by a herbivore. Land grant institutions and federal agencies have been conducting research on grazing management for the past 50–100 yr and many of the methods have become standardized (which is not always positive). Fire research, on the other hand, has lagged

Prescribed fire on sand sagebrush rangelands in Oklahoma that was intended to create heterogeneity for wildlife, variable grazing distribution, and control the invasion of Eastern Redcedar. (Photo: John Weir)

greatly, and has increased meaningfully in only the past decade. The lag in research is largely due to limited recognition of the importance of fire in a grazing-centric discipline and the concomitant limitation in research funding (partially alleviated by the Joint Fire Science Program [JFSP]). Our synthesis suggests that relevance of fire regime research for management goals continues to lag behind grazing regime research.

The use of fire to improve wildlife habitat is a complex issue that is not easily evaluated because some species respond positively and others respond negatively to prescribed fire. Therefore, fire-improved habitat for one species likely translates into fire-degraded habitat for another. Groups of organisms that are negatively influenced often recover rapidly unless they occur in a highly fragmented landscape where dispersal from unburned areas is limited. These conclusions have led some authors to suggest that heterogeneity should be promoted to maintain a shifting mosaic landscape so that the entire landscape is not burned or unburned at any point in time, but research is lacking to support this across most rangeland types. Moreover, research on wildlife response to prescribed burning has preferentially focused on birds and small mammals. This likely reflects the fact that these species groups are more sensitive to fire-altered habitat than most other wildlife, and that these species are more easily studied. Consequently, specific species and not merely species groups must be identified when developing burning programs to support wildlife.

The responses of soils and hydrology to fire are highly variable, but water repellency (i.e., hydrophobic soil) is a negative effect of extreme soil heating that occurs mostly under intense wildfire where fire has been absent for many years or in ecosystems with less evolutionary importance of fire. In general, fire increased water repellency, runoff, and sediment loads and decreased infiltration and water quality. Most of these effects disappeared after 2–4

Landscape photo of northern British Columbia demonstrating a shifting mosaic of fire patterns. Vegetation patterns are all due to variable times since fire resulting in highly variable plant communities. (Photo: Sam Fuhlendorf)

yr. Variables that influenced the effects of fire on hydrology were aspect, fire severity, and microsite characteristics. Soil chemistry is also highly variable and dependent on the ecosystem studied, pre- and postfire conditions, invasive species, and grazing. Some fire prescriptions, wildfires, fire-return intervals, and fuel situations in rangeland can result in extreme soil heating, which can markedly change soil chemical and physical properties. Otherwise, fire events corresponding to the evolutionary fire regime have short-lived and slight influence on soil chemistry.

RECOMMENDATIONS

1. The need for historic fire regimes to maintain the structure and composition of historic plant communities is well supported by ample scientific evidence. Many of the purposes in this conservation standard would benefit from the incorporation of more-specific goals and outcomes that can be more effectively compared to and supported by evidence in the peer-reviewed literature. Refer to Table 7 for evidence provided by the peer reviewed scientific literature supporting the current NRCS purposes for prescribed burning.
2. Conservation outcomes of prescribed burning are most likely to be attained when the specific purposes for prescribed burning are tailored to the unique characteristics of the ecosystems being managed. Highly generalized purposes are necessary for the establishment of national guidelines, but they may often be misleading because ecological processes such as seed production, seed germination, plant mortality, etc. are likely to be highly variable among ecoregions and even within communities and soils within ecoregions. Even widely accepted generalizations, such as the NRCS purpose that fire can be used to control undesirable vegetation, carry caveats and exceptions when details are considered.

3. Fire effects on ecosystems are often considered to be static over time, even though the research literature indicates that fire effects vary with time since fire and time between successive fires. Therefore, conservation purposes need to incorporate temporal dynamics to the extent that this information is available. Rangeland ecosystems evolved under specific fire regimes rather than in response to individual fires, which requires that conservation programs incorporate comprehensive fire regimes that address both short- and long-term outcomes. Ecosystem responses to fire and the effects of fire are both strongly influenced by temporal scale and must be carefully considered in conservation planning. Ecological site descriptions and rangeland research suggest that the prescribed burning standard should elevate application (i.e., area, number of plans, etc.) of prescribed burning so that it receives as much or more emphasis as the application of the conservation standard for grazing. There is no single practice as important to the maintenance of rangeland ecosystems.
4. Given the foregoing cautions against nationally generalized purposes, the following purposes for the practice of prescribed burning are supported by the research literature. These purposes should consider the context of the fire regime, rather than a single fire in isolation:

    a. to alter plant composition, reduce undesirable herbaceous plants, and reduce accumulated litter for a short period of time (generally 1 yr);
    b. to improve forage quality for < 3 yr;
    c. to limit encroachment of fire-sensitive shrubs and trees;
    d. to manage stature of resprouting, fire-adapted shrubs and trees;
    e. to alter distribution of grazing and browsing to either promote uniform distribution (by spatially homogenizing attractiveness) or to promote heterogeneity for biodiversity;
    f. to reduce potential for high intensity, stand-replacing fire by reducing accumulated fine fuel;
    g. to maintain grassland, savanna, and shrubland ecosystem structure, i.e., to prevent transitioning to woodland; and
    h. to recognize that mosaics of plant communities that vary with time since fire are critical for wildlife diversity.

KNOWLEDGE GAPS

1. Fire is as critical as climate and soils to the maintenance of ecosystem structure and function in many systems, but only limited experimental evidence exists to support the specific NRCS purposes for prescribed burning, especially those that involving long fire-return intervals.
2. The experimental literature supporting prescribed burning is in need of greater managerial relevance that can be obtained by directly addressing spatial scale, temporal scale, and interaction with other disturbances, including drought and grazing. Reliance on information resulting from single fires applied on small plots tracked for a relatively short time interval greatly constrains inferences and application to ecosystem management and this information should be applied with caution.
3. Knowledge of smoke characteristics and smoke management, effects of fire on wildlife and insects, and fire behavior and fire management exist in some regions, but are limited in other regions. The lack of sufficiently meaningful data on these topics makes it difficult to inform prescribed burning practices at the national level.

CONCLUSION

The vast majority of scientific evidence addressing fire in rangeland ecosystems points to the value of and need to continue or restore fire regimes with conservation programs. This is especially relevant given the accelerating rate of woody plant encroachment in grasslands and savannas, but it is important to other conservation outcomes, including altering grazing behavior of ungulates and maintaining biodiversity. The incorporation of prescribed burning conservation programs must include an understanding of the dynamic role that fire plays in most rangeland ecosystems. Fire regimes are equivalent to soils and climate in terms of their influence on plant community

Fire in a mixed grass prairie. (Photo: Steve Winter)

composition and landscape patterns. Recognition of fire as a dynamic regime that has variable effects depending on the interaction with grazing and climate patterns is critical to optimizing conservation effectiveness in rangeland ecosystems.

The complex interaction of scientific knowledge, social concerns, and variable policies across regions are major limitations to the successful and critical restoration of fire regimes. Successful grassroots actions that have led to increased use of prescribed fire include the development of prescribed fire cooperatives through many of the Great Plains states. These cooperatives build on regional strengths to bring landowners together to conduct prescribed fires on landscapes that have variable ownerships. These cooperatives have enabled landowners to overcome issues associated with labor, liability, and training and to restore fire regimes in regions that have had fire removed for more than a century. Landowner cooperatives have the potential to transform the application of fire and indicate that successful conservation practices based on fire are possible even in areas that do not have a history of prescribed burning.

To address the shortfalls in research applicable to prescribed burning on rangelands and the limited application of prescribed burning on rangelands, we recommend that the NRCS position itself to drive rangeland research and fire research agendas. Research to support NRCS purposes for prescribed burning on rangeland, unlike forests, has been limited by insufficient funding. However, a NRCS-driven research agenda also is lacking because NRCS has been detached from federal fire programs, notably LANDFIRE and the JFSP. Involvement in these programs would afford NRCS a greater opportunity to engage the scientific community and interact with other federal agencies that are developing valuable fire management tools and promoting fire research. This would provide NRCS with a platform for developing a research agenda relevant to supporting fire management on private lands.

LANDFIRE is an interagency vegetation, fire, and fuel characteristics mapping project (www.landfire.gov). Developed through cooperation of natural resource agencies other than NRCS (Bureau of Land Management, US Forest Service, National Park Service, etc.) and The Nature Conservancy, LANDFIRE was initiated by a request from federal agencies to develop maps needed to help land managers prioritize areas for hazardous fuel reduction and conservation actions. LANDFIRE has spatial data layers that include all layers required to run fire modeling applications such as FARSITE and FlamMap, existing vegetation type, canopy height, biophysical setting, environmental site potential, and fire regime condition class, as well as fire effects layers. Since its initiation, LANDFIRE has been expanded to address the entire United States, including private as well as public land, but it also has some data needs. Vegetation dynamics models that operate with Vegetation Dynamics Development Tool software form a major component of LANDFIRE. These state-and-transition models are similar to those being developed by NRCS in that they describe pathways of vegetation succession and the frequency and effects of disturbances; however, these models are quantitative and based on extensive field analyses and modeling. Currently, rangelands are underrepresented and could be enhanced by involvement of NRCS in this national fire effort. We recommend that the NRCS engage the development and use of decision tools like LANDFIRE and OK-Fire (2009) that facilitate management and application of fire on rangelands.

The JFSP was created by Congress in 1998 as an interagency research, development, and applications partnership between the US Department of the Interior and the US Department of Agriculture–Forest Service (http://www.firescience.gov/index.cfm). Funding priorities and policies are set by the JFSP Governing Board, which includes representatives from the Bureau of Land Management, National Park Service, US Fish and Wildlife Service, Bureau of Indian Affairs, and US Geological Survey, as well as five representatives from the Forest Service. This program funds applied research focused on the management of fire for natural resource managers. NRCS should work to become a JFSP partner to support research applicable to privately owned rangelands. This would provide NRCS with a platform for developing a research agenda relevant to supporting fire management on private lands.

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CHAPTER

3

Brush Management as a Rangeland Conservation Strategy: A Critical Evaluation

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“Woody plant encroachment represents a threat to grassland, shrub-steppe, and savanna ecosystems and the plants and animals endemic to them…”

Brush Management as a Rangeland Conservation Strategy: A Critical Evaluation

3

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INTRODUCTION

Rangelands support the majority of the world's livestock production (Safriel and Adeel 2005) and play an important role in human health and global carbon, water, and nitrogen cycles (Campbell and Stafford Smith 2000). Their extensive airsheds and watersheds provide habitat for game and nongame wildlife and myriad ecosystem goods and services important to rapidly growing settlements and cities that may be geographically distant. Rangelands thus have considerable, multidimensional conservation value. Stewardship of vegetation composition, cover, and production is the foundation of sustainable rangeland management, a key component of which is maintaining vegetation within a desirable mix of herbaceous and woody plants (WPs).

One of the most striking land cover changes on rangelands worldwide over the past 150 yr has been the proliferation of trees and shrubs at the expense of perennial grasses. In some cases, native WPs are increasing in stature and density within their historic geographic ranges; in other cases, nonnative WPs are becoming dominant. These shifts in the balance between woody and herbaceous vegetation represent a fundamental alteration of habitat for animals (microbes, invertebrates, and vertebrates) and hence a marked alteration of ecosystem trophic structure. In arid and semiarid regions, increases in the abundance of xerophytic shrubs at the expense of mesophytic grasses represent a type of desertification (e.g., Schlesinger et al. 1990; Havstad et al. 2006) often accompanied by reductions in primary production (Knapp et al. 2008a) and accelerated rates of wind and water erosion (Wainwright et al. 2000; Gillette and Pitchford 2004; Breshears et al. 2009). In semiarid and subhumid areas, encroachment of shrubs and trees into grasslands and savannas may have neutral to substantially positive effects on primary production, nutrient cycling, and accumulation of soil organic matter (Archer et al. 2001; Knapp et al. 2008a; Barger et al. 2011). While impacts of WP encroachment may vary among bioclimatic zones, there is one constant: grass-dominated ecosystems are transformed into shrublands, woodlands, or forest. As such, WP encroachment represents a threat to grassland, shrub-steppe, and savanna ecosystems and the plants and animals endemic to them, a threat on par with those posed by exurban and agricultural development (Sampson and Knopf 1994; Maestas et al. 2003).

Efforts to counteract the real and perceived threats of WP encroachment fall into the broad category of brush management. Brush management, defined by the Natural Resource Conservation Service (NRCS 2003) as the removal, reduction, or manipulation of nonherbaceous plants, has been an integral component of range management since its formal emergence in the 1940s. However, brush removal has historically been criticized, especially when large-scale programs have failed to consider the needs of diverse stakeholders and the impact on multiple goods and services during planning and implementation stages (e.g., Klebenow 1969; Belsky 1996).

Our goal here is to provide a contemporary, critical evaluation of "brush management" as a conservation tool. We begin with a brief review of potential drivers of WP encroachment. An understanding of these drivers will 1) shed light on the causes for the changes observed to date; 2) help us determine if management

Woody plant encroachment has been widespread in rangelands, including these desert grasslands in New Mexico. (Photo: Paolo D'Odorico)

TABLE 1. Potential causes for increases in woody plant (WP) abundance in rangelands. There is likely no single-factor explanation for this widespread phenomenon. Most likely, it reflects drivers that vary locally or regionally or from the interactions of multiple drivers. Changes in a given driver may be necessary to tip the balance between woody and herbaceous vegetation but may not be sufficient unless co-occurring with changes in other drivers. For detailed reviews and discussions, see Archer (1994), Archer et al. (1995), Van Auken (2000), Briggs et al. (2005), and Naito and Cairns (2011).

Driver	Mechanism	Potential vegetation response
Climate	Increased precipitation	Enhances WP establishment, growth, and density
	Decreased precipitation	Promote shifts from mesophytic grasses to xerophytic shrubs
	Shift from winter to summer precipitation	Favors WP over grasses, particularly on relatively deep, well-drained soils
Grazing	Utilization of grasses by livestock	Herbaceous production and species composition may shift to a community more susceptible to WP encroachment; livestock are effective agents of dispersal of some WP species; reductions in fine fuel mass and continuity (see "Fire")
	Seed dispersal	
Browsing	Reduced utilization of WPs by native herbivores	Elimination of browsers promotes WP recruitment and growth; WPs kept small in size by browsers more susceptible to fire
Fire	Reduced fire frequency, intensity, and extent	Increased WP recruitment and growth (see "Grazing")
Atmospheric CO_2	Increased atmospheric CO_2 concentrations	WPs with C_3 photosynthetic pathway may be favored over grasses with C_4 photosynthetic pathway
Nitrogen deposition	Increased N availability	Correlated with forest expansion into grassland

intervention is realistic; if so, 3) what approaches might be most effective; and 4) when, where, and under what conditions to apply them. We then discuss the ecological role of WPs in rangeland ecosystems and how human perspectives on WPs in rangelands influence management decisions and conservation objectives. The ecological impacts of WP proliferation are then reviewed with the aim of addressing the question, What are the environmental consequences of not managing WPs in rangelands? As it turns out, there are indeed consequences. Many of these have emerged relatively recently and hence are not yet reflected in current management guidelines. Advances in our understanding of the ecological consequences of WP proliferation in rangelands have paralleled changes in both perspectives on and approaches to brush management since the mid-1900s and have influenced how the NRCS has advised landowners. We therefore review the evolution of brush management in the spirit of putting current perspectives into their historical context. The basis for NRCS expectations underlying recommendations in the NRCS Brush Management Conservation Practice Standard matrix (hereafter described as "projected effects") is then evaluated on the basis of a pooling of expectations into five overarching areas: herbaceous cover, production, and diversity; livestock response; watershed function; wildlife response; and fuels management. Evaluations are then followed by recommendations, an itemization of knowledge gaps, and a series of overarching conclusions.

WHY HAS WP ABUNDANCE INCREASED ON RANGELANDS?

Understanding the drivers of tree/shrub encroachment can help identify when, where, how, and under what conditions management might most effectively prevent or reverse WP proliferation. Traditional explanations center around intensification of livestock grazing, changes in climate and fire regimes, the introduction of nonnative woody species, and declines in the abundance of browsing animals (Table 1). Historical increases in atmospheric nitrogen deposition and atmospheric carbon dioxide concentration are also potentially important drivers. Exploring this important question is beyond the scope of this discussion, but detailed reviews and discussion can be found in Archer (1994), Archer et al. (1995), Van Auken (2000), Briggs et al. (2005), and Naito and Cairns (2011). Likely all these

factors have interacted to varying degrees, and the strength and nature of these interactions likely varies from one biogeographic location to another. Thus, local knowledge is important in developing WP management plans. In many respects, WP encroachment is a specific case of weed and invasive plant management, and the concepts and principles developed for those perspectives are widely applicable (Sheley et al. this volume).

It is important to note that once the process of WP encroachment is set in motion, grazing management per se may do little to prevent the conversion of grasslands and savannas to shrublands and woodlands (e.g., McClaran 2003; Browning et al. 2008). In fact, on sites with a long history of heavy grazing, removal of livestock may actually promote rather than deter WP encroachment (Smeins and Merrill 1988; Browning and Archer 2010). However, grazing management influences on WP encroachment are indirectly important in terms of how they affect the amount and continuity of fine fuels available for wildfire or prescribed burning (Fuhlendorf et al. 2008; Fuhlendorf et al. this volume). Because grazing management alone is generally not sufficient to curtail or reverse shrub encroachment, progressive brush management is a potentially important tool for grassland conservation.

Although WP encroachment has been formally documented and qualitatively observed in some areas, it should not be assumed that this transformation has been uniform or ubiquitous. Indeed, repeat ground photography in western North America documents areas where WPs have dominated landscapes since the 1800s (e.g., Humphrey 1987; Turner et al. 2003; Webb et al. 2007). Thus, many areas may have been historically comprised of mixtures of woody and herbaceous vegetation (e.g., shrub-steppe or shrub or tree savannas), and efforts to eradicate WPs from such sites may be misguided (McKell 1977) and sometimes detrimental to native plants and wildlife (e.g., Knick et al. 2003).

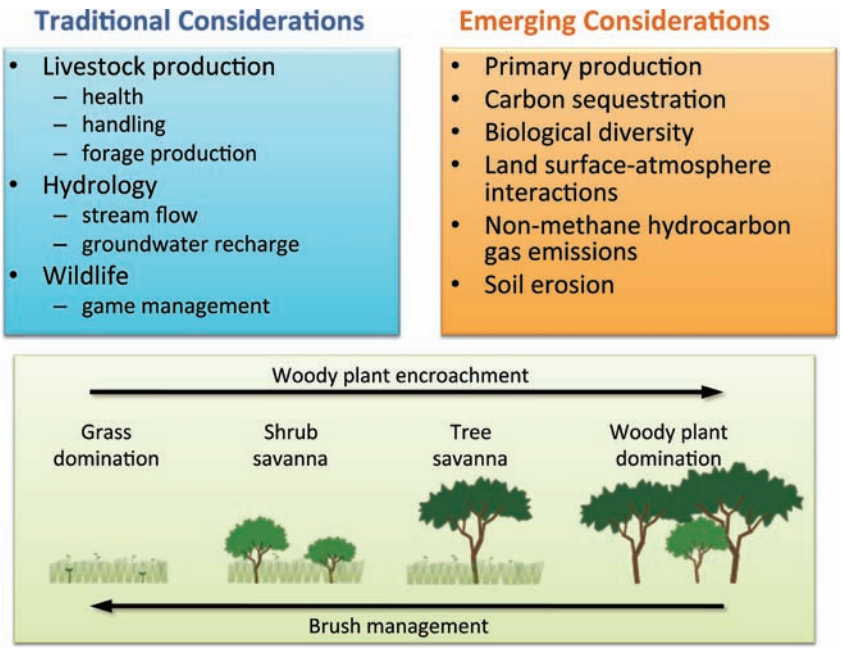
PERSPECTIVES ON WPs IN RANGELANDS

Brush management practices have historically focused on the goal of maximizing livestock production and promoting groundwater recharge and stream flow. Contemporary perspectives have been broadened to include impacts on biological diversity, ecosystem function (primary production and nutrient cycles), and land surface–atmosphere interactions (Appendix 2; Fig. 1). These broader perspectives are recognized to varying levels of specificity in NRCS Brush Management Conservation Practice Standards (code 314) and its projected effects. The current challenge lies with articulating these more explicitly in the CPPE worksheet, exposing landowners and the public to these perspectives, and articulating these perspectives in terms that can be quantified and objectively monitored and evaluated.

TABLE 2. Perspectives on woody plants (WPs) in rangelands. In areas subject to heavy livestock grazing, palatable species typically give way to less palatable, less preferred species, and in rangelands, these less palatable species are often shrubs. The fact that unpalatable shrubs dominate many grazed rangelands has led to the mistaken generalization that all WPs in rangelands are undesirable. WPs have been typically viewed as the problem on grazed rangelands, but in fact they are likely a consequence of past mismanagement. Brush management conducted in isolation of grazing management is therefore treating symptoms rather than addressing the root causes of the problem (excessive grazing and fire suppression). When assessing whether to invest in efforts to reduce WP cover or density, the points shown in the table should be considered. For further discussion, see McKell (1977), Archer and Smeins (1993), and Archer (2009).

• Palatable WPs may have been displaced along with palatable grasses and herbs (Lange and Willcocks 1980; Orodhu et al. 1990; Kay 1997).
• Shrubs may decrease grazing pressure on grasses and provide protection for heavily utilized herbaceous species.
• WPs may provide important habitat for a variety of vertebrate and invertebrate wildlife (nongame as well as game).
• Shrubs may provide an important and underappreciated source of nutrimental stability and reduce supplemental feed requirements during cold or dry periods (Le Houérou 1980; Coppock et al. 1986; Stuth and Kamau 1990; Styles and Skinner 1997).
• WPs may be the best adapted for the prevailing environmental conditions (Le Houérou 1994).
• Were it not for the "damn brush," there might be little or no vegetative cover. It may not be realistic to expect brush management to enhance herbaceous production, especially where soils have extensively eroded.
• Will brush management stimulate herbaceous production and increase livestock carrying capacity sufficiently to offset treatment costs? If so, how much time will be required before a follow-up treatment? Will treating one problem perhaps create another (i.e., loss of valuable nontargeted species, invasion by weeds or exotic species, induced multiple-stemmed growth habit in shrubs, or replacement of nonsprouting species with sprouting species)?

FIGURE 1. Traditional and contemporary perspectives on woody plant (WP) encroachment (from Archer 2009).



THE COST OF DOING NOTHING

Changes in WP cover and density represent fundamental changes in vegetation composition and structure and animal (microbes, invertebrates, and vertebrates) habitat. These, in turn, can fundamentally alter ecosystem primary production, trophic structure, biological diversity, nutrient cycling, and land surface–atmosphere interactions (Fig. 2).

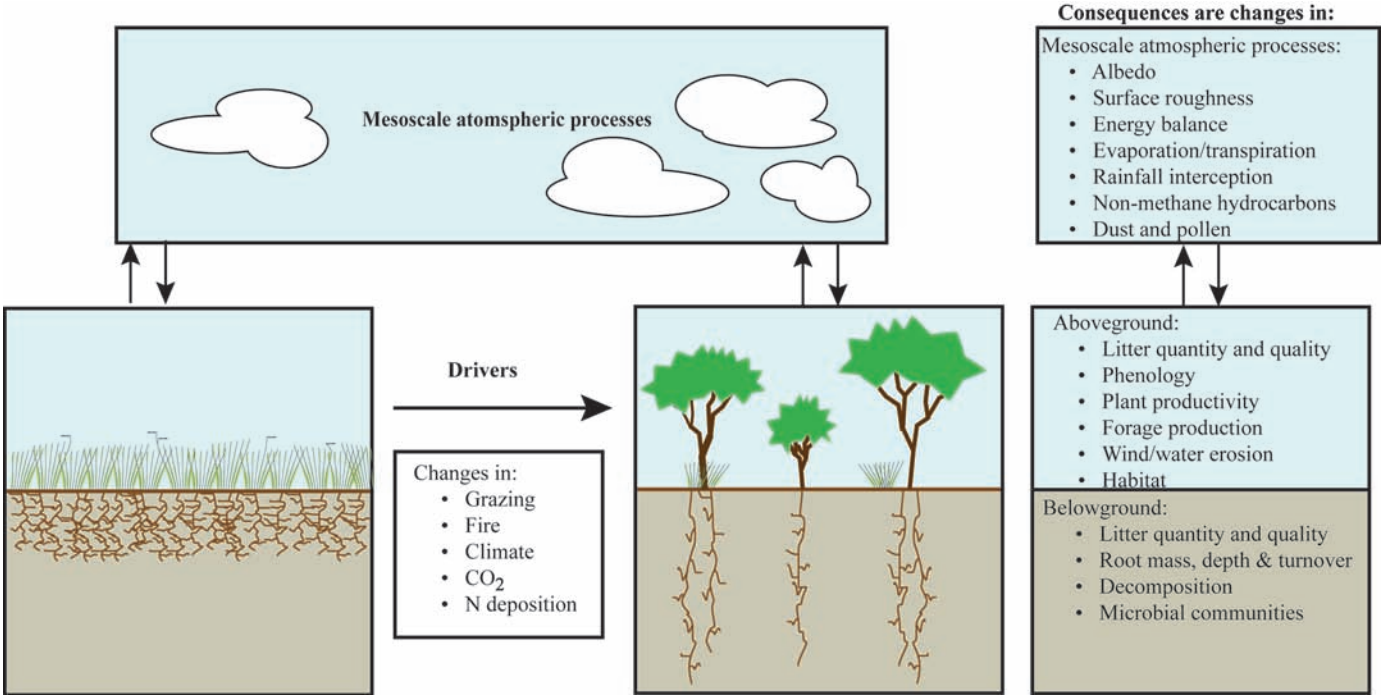
Herbaceous Cover and Production

The projected effects of brush management typically assume that herbaceous cover and production will increase following brush management. Implicit in this expectation is the assumption that WPs have a negative impact on ground cover. Does the literature support this perspective? The answer to this question is context dependent. Herbaceous cover and biomass typically decline as WP cover and basal area increase. However, the specific nature of the response ranges from an immediate linear or exponential decline to an initial stimulation, followed by a subsequent decline (Fig. 3; Table 3). The shape of these curves depends on the site and its grazing history, its climate, the physiology of the herbaceous vegetation (e.g., cool-season C_3 vs. warm-season C_4 grass), and the species of WP and its growth form (i.e., evergreen vs. deciduous), canopy architecture (i.e., single vs. multiple stemmed), size, density, and spatial arrangement (Jameson 1967; Mitchell and Battling 1991; Scholes and Archer 1997; Scholes 2003; Fuhlendorf et al. 2008; Teague et al. 2008a). When stocking rates are based on total area rather than grazable area, WP encroachment can intensify grazing pressure to further depress grass production unless stocking rates are adjusted to compensate for WP-induced losses of forage production. WP impacts on herbaceous plants must therefore be considered in the context of livestock management (Briske et al. this volume) and the ecological site(s) being managed.

It is important to note that relationships between WPs and grass are typically described at either the plant or the stand scale of spatial resolution. This can cause confusion and apparent contradictions. For example, velvet mesquite (*Prosopis velutina*) typically has neutral to positive effects on grasses at the plant scale but negative effects at the stand scale (McClaran and Angell 2007). Thus, care should be taken when generalizing results from a given study. Stand-scale assessments are generally most appropriate for pasture and landscape management.

How do declines in herbaceous cover and biomass that typically accompany WP encroachment impact overall ecosystem primary production? A recent comparison of sites around North America suggests aboveground primary production declines with WP encroachment in hot and cold deserts but that it increases dramatically as a function of annual rainfall in semiarid and subhumid regions (Knapp et al. 2008a). Recent estimates suggest that for every millimeter increase in mean annual precipitation above 330 mm, aboveground net primary production (ANPP) will increase by ~0.6 g $C \cdot m^{-2} \cdot yr^{-1}$ with shrub encroachment (Barger et al. 2011). Thus, losses of grass production can lead to a net decline in overall ecosystem production in arid areas, whereas increases in production attributable to WPs more than compensates for declines in herbaceous production in other bioclimatic zones.

FIGURE 2. Drivers of woody plant encroachment (see Text Box 1) and the potential consequences of ecosystem function and land surface–atmosphere interactions (from Archer 2009).



Soil Condition and Erosion

Projected effects generally assume that soil conditions and soil surface stability will be slightly to substantially improved by brush management and that soil erosion will be reduced by WP removal. Although not explicitly stated, these assumptions appear predicated on the expectation that WP proliferation adversely affects these parameters. Does WP encroachment lead to a deterioration of soil condition and site stability?

Changes in grass and WP abundance impact soils through alteration of above- and belowground productivity, quality of litter inputs, rooting depth and distribution, hydrology, microclimate, and energy balance (Fig. 2). The abundance of soil organic matter or, more precisely, soil organic carbon (SOC) is a good indicator of soil condition, as it integrates a variety of ecosystem processes that influence fertility, water-holding capacity, and site stability.

A substantial majority of the carbon in rangeland ecosystems resides in the SOC pool (Schlesinger 1997), but it is not yet clear how grazing, climate, and WP encroachment and "infilling" (shifts from relatively low to relatively high WP cover or density) interact to affect gains and losses from these large carbon pools. Despite consistent increases in aboveground carbon storage with woody vegetation encroachment (Knapp et al. 2008a) and dryland afforestation (e.g., Nosetto et al. 2006), the trends in SOC are highly variable, ranging from substantial losses to large gains to

FIGURE 3. Potential responses of herbaceous vegetation to increases in woody plant cover or basal area. See Table 1 and reviews by Jameson (1967), Mitchell and Battling (1991), Scholes and Archer (1997), Scholes (2003), Fuhlendorf et al. (2008), and Teague et al. (2008a).

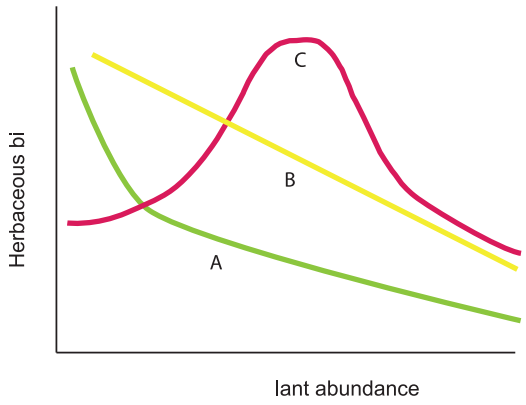


TABLE 3. Herbaceous response to shrub encroachment (US studies only).

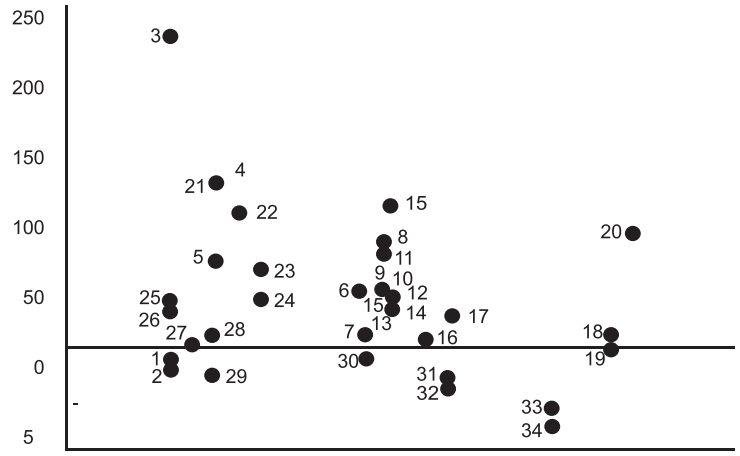
Dominant woody plant(s)		Herbaceous response[a]	Soils	MAP (mm)	
Big sagebrush (*Artemisia tridentata*)		D[1]	Aeolian sandy loams and loess (aridisols)	210	
		A	Loam/sandy loam	225–405 (eight sites)	
		D[2]	Silt loam	406	
		C	—	279	
Broom snakeweed (*Gutierrezia sarothrae*)		A	Gravelly loam	323 (Vaughn) 328 (Roswell)	
		B	Fine sandy loam	480	
Creosotebush (*Larrea tridentata*)		B, D[1]	Sandy loam	221–430 (4 sites)	
		D[1]	Gravelly/loamy	240	
		A/B	Shallow sandy		
		C	—	255	
		A	Gravelly	—	
Honey mesquite (*Prosopis glandulosa*)		C	Clay loam	—	
		A	Sandy loan	231	
		B	Silt loam/clay loam	648	
		B	Loamy sand	231	
		B	Clay loams	665	
		D[2]	Shallow clay		
		C	Sandy loam	345	
Huisache (*Acacia farnesiana*)		B (total); C (cool-season grass)	Clay	850	
Juniper (*Juniperus* spp.)	**Redberry juniper (*J. pinchottii*)**	B	—	—	
	Redberry juniper (*J. pinchottii*)	A (grazed); B (ungrazed)	Fine loam and clay	~525	
	Western juniper (*J. occidentalis*)	B	Loams with variable rockiness	410	
		D[2]	Clay loam	430	
		C	—	248	
Pine (*Pinus* spp.)	**Ponderosa pine (*P. ponderosa*)**	A	—	—	
	Ponderosa pine (*P. ponderosa*)	A	Limestone derived	560	
	Ponderosa pine (*P. ponderosa*)	A	—	380–510 (31 sites, varies with elevation)	
	P. taeda, P. echinata	A	—	—	
	Longleaf pine (*P. palustris*)	B	Silty loam	~584	
Pinyon–juniper (*Pinus edulis–Juniperus* spp.)		A	—	—	
		D[3]	—	345	
Velvet mesquite (*Prosopis velutina*)		D[2]	—	~330–432 (varies with elevation)	
		C (one site); D[2] (three sites)	—	~197–304 (varies with elevation)	
		A	—	—	

[a] As per Figure 3 (A, negative exponential decline; B, linear decline; C, initial positive response, followed by decline; and "D," "Other," including no change (D1), increase (D2), or decline in C4, increase in C3 (D3).

*Space-for-time substitution (sampling stands of different shrub abundance at one point in time).

	Scale (plant/ stand)	Study duration (yr)	Study location	Reference
	Plant	28 (1950–1978)	Southeastern ID	Anderson and Holte (1981)
	Stand	20	Northwestern NM	McDaniel et al. (2005)
	Stand	4 (1967–1970)	Southern ID	Hull and Klomp (1974)
	Stand	4 (1951–1954)	OR	Hyder and Sneva (1956)
	Stand	10 (1979–1989)	Vaughn and Roswell, NM	McDaniel et al. (1993)
	Stand	2 (1946–1947)	TX	Ueckert (1979)
	Stand	2* (1981–1982)	Chihuahua, Mexico, and southern AZ	Morton et al. (1990)
	Stand	14–17 (1984/1987–2001)	Southwest NM	Perkins et al. (2006)
	Stand	76 (1915–2001)	Central NM	Baez and Collins (2008)
	Stand	Model	South-central NM	Bestelmeyer et al. (2009)
	Stand	5 (1970–1975)	West TX	Dahl et al. (1978)
	Stand	105 (1858–1963)	South-central NM	Buffington and Herbel (1965)
	Stand	6 (1995–2001)	North-central TX	Teague et al. (2008a)
	Stand	45 (1935–1980)	Southern NM	Hennessy et al. (1983)
	Stand	1*1998	North-central TX	Hughes et al. (2006)
	Stand	46 (1957–2003)	Southern AZ	McClaran and Angell (2006)
	stand	2 (1978–1979)	South TX	Scifres et al. (1982)
	Stand	1*1995	Nolan County, TX	Johnson et al. (1999)
	Stand	3 (1984–1986)	Western TX	McPherson and Wright (1990)
	Stand	2* (2005–2006)	Northeastern CA	Coultrap et al. (2008)
	Stand	7 (1975–1982)	CA	Evans and Young (1985)
	Stand	13 (1991–2004)	Southeastern OR	Bates et al. (2005)
	Tree	1*	Northern AZ	Jameson (1967)
	Stand	1* (1988)	Northern AZ	Moore and Deiter (1992)
	Stand	1* (1984)	Rocky Mountain front range, CO and WY	Mitchell and Battling (1991)
	Stand	2* (1960–1961)	Eastern TX	Halls and Schuster (1965)
	Stand	11	Alexandria, LA	Grelen and Lohrey (1978)
	Stand	1*	Northern and central AZ	Jameson (1967)
	Stand	2 (1984–1985)	South-central NM	Pieper (1990)
	Stand	10 (nonsequential; 1954–1967)	Southern AZ	Cable (1971)
	Stand	5 (1945–1950)	Southern AZ	Parker and Martin (1952)
	Stand	—	Southeastern AZ	Upson et al. (1937)

FIGURE 4. Changes in soil organic pools following shrub encroachment in different precipitation zones (from Asner and Archer 2009). Numbers on symbols/legend entries indicate references listed in Appendix III.



no net change (Wessman et al. 2004; Asner and Archer 2009; Fig. 4).

Variation in SOC response to WP encroachment is perhaps not unexpected given the myriad factors that influence SOC pool and fluxes (Wheeler et al. 2007). These include growth characteristics of the WPs (e.g., evergreen or deciduous, N fixing or not, shallow or deep rooted, etc.), climate (mean annual rainfall and temperature), soil properties (e.g., texture, pH, carbonate content), initial conditions (e.g., amount, type, and distribution of SOC present at the time WP encroachment begins), and prior land management (e.g., whether WPs are establishing in native rangeland vs. abandoned cropland). In areas where shrub-induced increases in SOC have been documented, changes are typically restricted to the upper 10–20 cm of the soil profile, and accumulation appears to be a linear function of time since WP establishment (Boutton et al. 2009) with rates ranging from 8 g $C \cdot m^{-2} \cdot yr^{-1}$ to 30 g $C \cdot m^{-2} \cdot yr^{-1}$ (Wheeler et al. 2007). Some of the uncertainty in SOC response may reflect the fact that WP encroachment often occurs in areas with a history of livestock grazing, which itself has positive, neutral, and negative effects on SOC pools (Milchunas and Lauenroth 1993; Piñeiro et al. 2010; Briske et al. this volume). Where historical grazing and WP encroachment effects on SOC have been explicitly accounted for, it appears that losses of SOC associated with heavy grazing can be recovered subsequent to WP encroachment and that SOC in the shrub-dominated system can be substantially greater than that of the original grasslands (Archer et al. 2001; Hibbard et al. 2003).

SOC and N levels are typically highly correlated; hence, increases in SOC are typically accompanied by increases in soil N (Seastedt 1995; Wheeler et al. 2007). When the encroaching woody species is a nitrogen fixer, soil N levels can increase substantially (Geesing et al. 2000; Hughes et al. 2006), thus augmenting a key resource that, along with water, typically colimits rangeland productivity. The resultant increase in soil fertility and water-holding capacity likely drives the increase in herbaceous production that typically follows WP removal.

In arid regions, the loss of grass cover due to grazing is accompanied by loss and redistribution of soil resources from plant interspaces to areas beneath shrubs. Many studies have investigated this grass-erosion feedback, with the consensus that erosion by wind and water is capable of removing soil resources required for grass growth and propagation while creating semipermanent fertile islands beneath shrub canopies (see Okin et al. 2009). The net result is a dramatic increase in wind and erosion resulting from increased bare areas in shrublands compared to the grasslands they replaced. Aeolian sediment flux in mesquite-dominated shrublands in the Chihuahuan Desert are 10-fold greater than rates of wind erosion and dust emission from grasslands on similar soils (Gillette and Pitchford 2004). Flow and erosion plots in the Walnut Gulch Experimental Watershed in Arizona and the Jornada Long Term Ecological Research site in New Mexico have demonstrated significant differences in water erosion between grasslands and shrublands (Wainwright et al. 2000). For example, higher splash detachment rates (Parsons et al. 1991, 1994) and interrill erosion rates (Abrahams et al. 1988) are observed in shrublands compared to grasslands, and shrubland areas are more prone to develop rills, which are responsible for significant increases in overall erosion rates (Luk et al. 1993). Episodes of water

erosion are often associated with decadal drought–interdrought cycles because depressed vegetation cover at the end of the drought makes the ecosystem vulnerable to increased erosion when rains return (McAuliffe et al. 2006). In hot desert systems where shrub encroachment has occurred, reestablishment of grass cover would help curtail erosion losses. However, the loss of topsoil to date, coupled with low and highly variable precipitation, make these among the most challenging environments in which to reestablish perennial grass cover once it has been lost (see the section "Herbaceous Vegetation and Native Communities").

Air Quality and Land Surface–Atmosphere Interactions

Brush management impacts on air quality are treated as "not applicable" or "neutral" with respect to particulate matter, ozone, and greenhouse gas production. However, these factors and others related to pollen production and land surface–atmosphere interactions may warrant more attention in the next generation of projected effects worksheets.

A synthesis of aeolian sediment transport studies spanning a grassland–forest continuum suggests 1) that among relatively undisturbed ecosystems, arid shrublands have inherently greater aeolian transport because of wake interference flow associated with intermediate levels of density and spacing of WPs and 2) that among disturbed ecosystems, the upper bound for aeolian transport decreases as a function of increasing amounts of WP cover because of the effects of the height and density of the canopy on airflow patterns and ground cover associated with WP cover (Breshears et al. 2009).

Pollen from WPs trigger nasal allergies and asthma (Chang 1993; Gutman and Bush 1993). Tree/shrub proliferation thus has the potential to influence the onset, duration, concentration, and total production of pollen allergens both locally and at great distances (Levetin 1998). However, the role of these allergens on human health is not well understood (Al-Frayh et al. 1999).

Climate and atmospheric chemistry are directly and indirectly influenced by land cover via biophysical and biogeochemical aspects of land surface–atmosphere interactions. Shifts from grass to WP domination have the potential to influence biophysical aspects of land–atmosphere interactions related to albedo, evapotranspiration, surface roughness, boundary layer conditions, and dust loading that affect cloud formation and rainfall (Figs. 1 and 2). Increases in C and N pools that occur when WPs proliferate in grasslands and savannas may be accompanied by increases in trace gas emissions (e.g., carbon dioxide, nitrous oxide, and methane; McCulley et al. 2004; Sponseller 2007; McLain et al. 2008) and nonmethane hydrocarbon emissions (Monson et al. 1991; Guenther et al. 1995; Klinger et al. 1998; Geron et al. 2006). Emissions of such compounds can influence atmospheric oxidizing capacity, heat retention capacity, greenhouse gas half-life, aerosol burdens, and radiative properties. As a result, air quality (Monson et al. 1991) and energy balance can be affected.

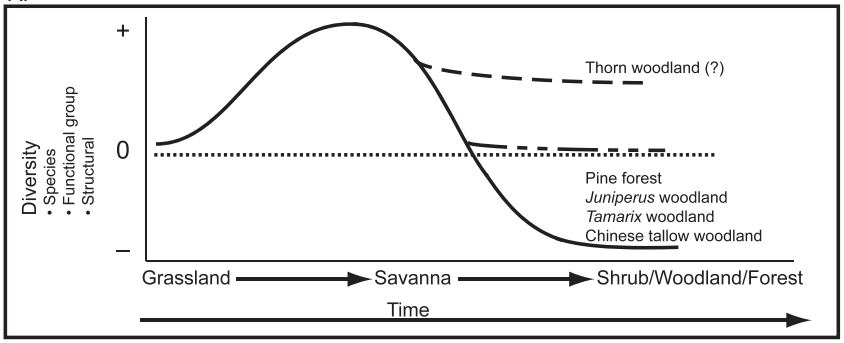
WP encroachment has been accompanied by increased dust production in arid regions (Gillette and Pitchford 2004). Dust can potentially influence weather and climate by scattering and absorbing sunlight and affecting cloud properties, though the overall effect of mineral dust in the atmosphere is likely

> "Woody plant encroachment has been accompanied by increased dust production in arid regions"

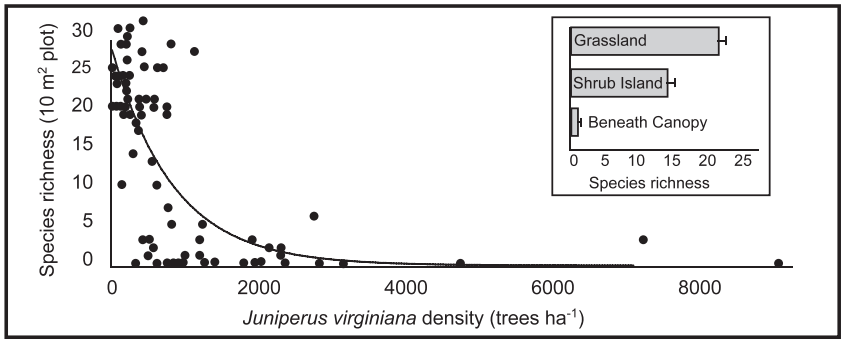
Changes in vegetation have the potential to influence biophysical aspects of land–atmosphere interactions via effects on trace gas and hydrocarbon emissions, albedo, evapotranspiration, surface roughness, and dust loading. (Photo: Rich Reynolds)

FIGURE 5. (**A**) Conceptual model of landscape-scale changes in ecosystem biodiversity (species, growth form, or structural) that potentially accompany woody plant (WP) proliferation in grasslands and savannas (from Archer 2009). Plot-scale reductions in herbaceous species richness with increases in (**B**) juniper (*Juniperus virginiana*) density (10-m^2 plots) and within *Cornus drummondii* shrub islands and surrounding grasslands (m^2 plots; insert) in Tall Grass Prairie (Knapp et al. 2008b) and (**C**) with creosote bush encroachment in Desert Grasslands (Baez and Collins 2008). In **A**, diversity is predicted to increase during early stages of WP encroachment because of the mixture of woody and herbaceous floral/faunal elements. Maximum diversity might be expected in savanna-like configurations where woody and herbaceous plants co-occur. As WP abundance increases, loss of grassland components eventually occurs. In subtropical thorn woodland and dry forests with high WP species richness, a net increase in diversity may result. In other settings, there may be no net change in diversity, only a change in physiognomy. Where WPs form virtual monocultures with little or no understory (e.g., panels **B** and **C**), the loss of diversity may be profound. Regardless of the numerical changes in biodiversity, the existence of grassland and open savanna ecosystems and the plants and animals endemic to them are jeopardized with WP encroachment.

A.



B.



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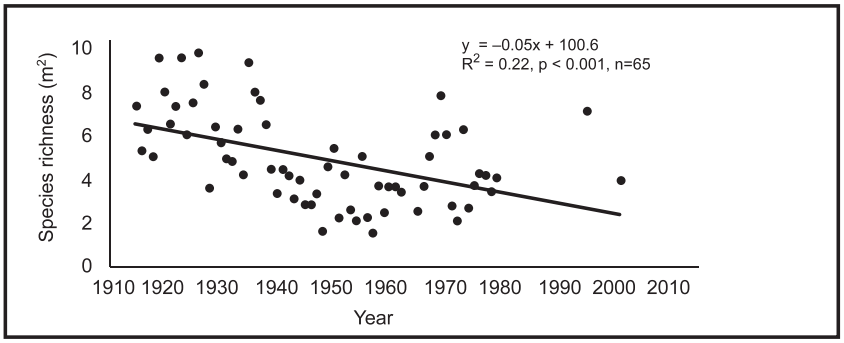


small compared to other human impacts (Intergovernmental Panel on Climate Change 2007). However, the mineral aerosols in dust originating from drylands are thought to play a major role in ocean fertilization and CO_2 uptake (Blain et al. 2007), terrestrial soil formation and nutrient cycling (Chadwick et al. 1999; Neff et al. 2008), and public health (e.g., Mohamed and El Bassouni 2007). Dust deposition also decreases albedo of alpine snowpack, thus accelerating melt and reducing snow-cover duration (Painter et al. 2007). In arid regions, erosion increases sediment delivery and changes flow conditions of rivers (Jepsen et al. 2003) and impacts water quality, riparian vegetation, aquatic fauna (Cowley 2006), and soil fertility and ecosystem processes (Valentin et al. 2005; Okin et al. 2006). Thus, the replacement of grasslands by shrublands in arid areas has potentially far-reaching ramifications.

Modified land cover can affect weather and climate (Bryant et al. 1990; Pielke et al. 1998). Changes in vegetation height and patchiness that occur when WPs replace grasses over large areas affect boundary layer conditions and aerodynamic roughness; changes in leaf area and rooting depth alter inputs of water vapor via transpiration; and changes in fractional ground cover, phenology, leaf habit (e.g., evergreen vs. deciduous), albedo, and soil temperature influence evaporation and latent and sensible heat exchange (Fig. 2; e.g., Graetz 1991; Bonan 2002). The extent to which these changes in structure influence meteorological conditions likely vary with annual rainfall (e.g., via leaf area changes accompanying shrub encroachment [Knapp et al. 2008a]), soil texture, shrub rooting depth, and proximity to water tables (Jobbagy and Jackson 2004).

Effects of WP encroachment on mesoscale climate and local weather have not been assessed. However, evidence from tree-clearing studies suggest that decreases in WP cover can potentially influence evapotranspiration, the incidence of convective storms, and cloud formation (Jackson et al. 2007). Model simulations in tropical savannas indicate that clearing of woody vegetation could increase mean surface air temperatures and wind speeds, decrease precipitation and humidity, and increase the frequency of dry periods within the wet season (Hoffman and Jackson 2000). Thus,

by extension, we might expect increases in WP abundance to have the reverse effect on local weather and climate.

Biodiversity

Effects of WP encroachment on biodiversity, whether quantified as the genetic diversity of populations, species richness, or the number of plant functional groups or animal guilds represented in an area, have not been widely quantified. At the landscape scale, colonization of grasslands by WPs initially represents new species additions and hence promotes biodiversity, and shrub modification of soil properties, vertical vegetation structure, and microclimate may subsequently promote the ingress and establishment of other plant and animal species (Fig. 5A). In its early stages, WP encroachment may have a multiplier effect on animal diversity by adding keystone structures and habitat heterogeneity (Tews et al. 2004b) and providing nesting, perching, and foraging sites and shelter against predators and extreme climatic conditions (Whitford 1997; Cooper and Whiting 2000; Valone and Sauter 2005; Blaum et al. 2007a). Indeed, numerous reptiles, birds, and mammals appear to prefer heterogeneous grass-dominated landscapes where scattered WPs provide up to 15% cover (Solbrig et al. 1996; Meik et al. 2002; Eccard et al. 2004; Bock et al. 2006; Thiele et al. 2008). In arid savanna rangelands, the diversity of small carnivores and their prey peaks at about 10–15% shrub cover (Blaum et al. 2007d). In the Chihuahuan Desert, shrub-invaded sites harbor four times the number of ant forager species found at a relatively pristine desert grassland site, suggesting that ant diversity is enhanced by shrub invasion and that several taxa benefit from it (Bestelmeyer 2005). The effects of WP encroachment vary among animal taxa and functional groups (e.g., Kazmaier et al. 2001), but as WP cover increases and habitat characteristics continue to shift, shrubland/woodland-adapted species are expected to become favored over grassland-adapted species.

> “Grassland-obligate plants and animals may be affected immediately and negatively by WP encroachment”

Grassland-obligate plants and animals may be affected immediately and negatively by WP encroachment (Table 4). Even so, diversity may be maintained or enhanced if new species co-occur with the more broadly adapted original species and if the displacement of grassland-obligate species is more than offset by the arrival of new species (e.g., Sauer et al. 1999; Blaum et al. 2007b, 2007c). As WP cover and biomass continue to increase, the end result may be an overall gain in diversity, no net change in diversity, or a net loss in diversity (Fig. 5A). Qualitative observations suggest that tropical and subtropical grasslands may potentially experience net gains in diversity with WP encroachment because of large pools of tree and shrub species, large pools of herbaceous species capable of coexisting with WPs, and large pools of invertebrates

TABLE 4. Avifauna and woody plant (WP) encroachment. Grassland passerines are declining at a faster rate than any other bird group in North America (Peterjohn and Sauer 1999). Woody plant encroachment associated with livestock grazing is among the contributing factors (Bakker 2003; Brennan and Kuvlesky 2005).

Vegetation change	Effects on grassland avifauna	Citation(s)
Thresholds of WP cover and height exceeded	Reduction in suitable habitat	Lloyd et al. (1998); Grant et al. (2004); Gottschalk et al. (2007)
Proximity to woodlands	Decreased food abundance; increased predation and brood parasitism	Johnson (2001); Bakker et al. (2002); Fletcher and Koford (2002); Thiele et al. (2008)
Grasslands converted to shrublands	Increased overall avian species richness but declines in ground-nesting passerine and gallinaceous species	Whitford (1997); Pidgeon et al. (2001); Rosenstock and van Riper (2001); Fuhlendorf et al. (2002)
Juniper encroachment into sagebrush–steppe communities	Eliminates sagebrush-obligate species habitat	Connelly et al. (2000); Miller et al. (2000); Crawford et al. (2004); Reinkensmeyer et al. (2007)

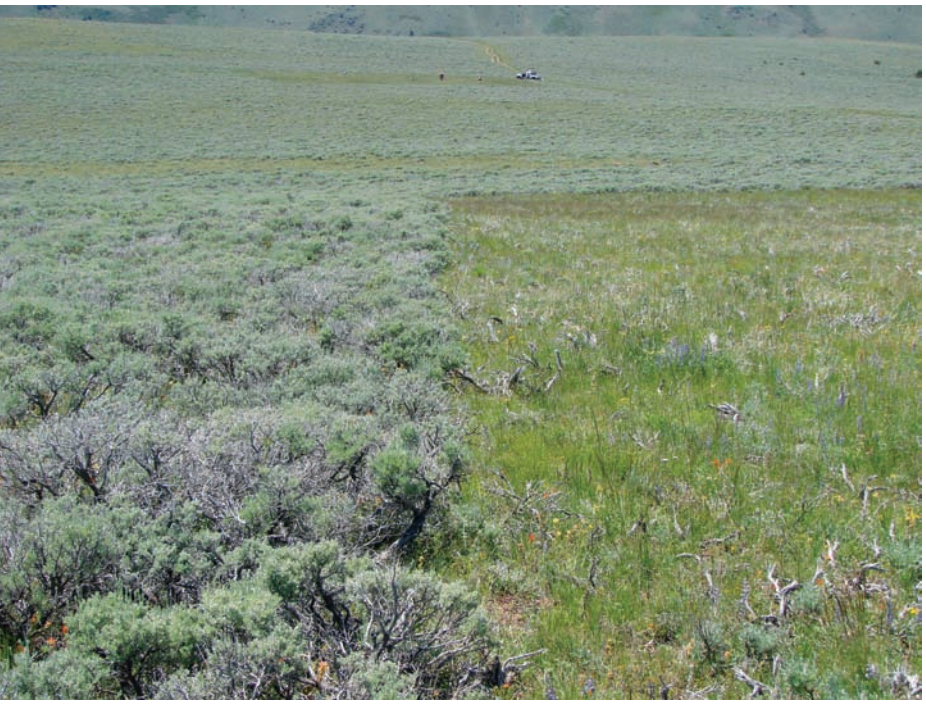


FIGURE 6. Undisturbed (left) and mowed (right) mountain big sagebrush (*Artemisia tridentata vaseyana*)–bunchgrass plant community in southeastern Oregon. (Photo: K. W. Davies)

and passerine bird species. In other cases, the number of encroaching woody species may be very small and their traits detrimental to the persistence of other plant species. WP encroachment may then result in virtual monocultures of vegetation (Figs. 5B and 5C) with concomitant impacts on faunal diversity.

Changes in aboveground biological diversity with WP proliferation may also be accompanied by changes in the diversity of microbial communities belowground. Shifts from bacterial to fungal populations may accompany shifts from herbaceous to woody domination (e.g., Imberger and Chiu 2001; Purohit et al. 2002), enabling the microbial biomass to effectively deal with lower litter quality and thus maintain or even increase soil respiration and mineralization. Aanderud et al. (2008) found differences in gram-positive bacteria, Actinobacteria, and fungi communities in soils below and between shrubs. Thus, changes in microbial communities would be expected to accompany changes in composition and abundance of shrubs.

Parasitic nematodes and nematodes feeding on bacteria and fungi in the immediate vicinity of plant roots are indicator taxa for changes in belowground microbial communities. The maximum depth of occurrence of these organisms increased from 2.1 m in grasslands to 4.0 m in areas where WPs have replaced grasses, but the composition of the nematode food web at this depth was markedly reduced from five trophic groups to two (Jackson et al. 2002). Invaded woody sites also had lower species richness in soils due primarily to the loss of root feeding species.

The conceptual model in Figure 5A is based on numerical assessments of species, functional group, and structural diversity. However, from the perspective of physiognomic diversity, WP encroachment is transformative. Grasslands become shrub or tree savannas, and shrub and tree savannas become shrublands or woodlands. Thus, even in cases where numerical diversity may be maintained or enriched by WP encroachment, there is a loss of grassland and savanna ecosystems and the plants and animals endemic to them. Thus, while brush management has historically been advocated from the perspective of potential benefits for livestock production and hydrology, it should also be considered from the perspective of maintaining the existence of grassland and savanna ecosystems.

BRUSH MANAGEMENT: A BRIEF HISTORY

WP encroachment has long been of concern to rangeland managers (Leopold 1924). Thus, there is a long history of devising management tools for reducing WP cover. The basis for concerns over WP proliferation was historically centered around the adverse effects of shrubs on forage production (Fig. 6) and livestock safety (e.g., WPs as cover for predators), health (e.g., as habitat for insect and arthropod pests and parasites such as ticks and horn flies [Teel et al. 1997]), and handling (difficulty in gathering and moving animals with increasing WP stature/cover/density). This traditional focus on rangeland value for livestock production was also the impetus for other management practices, such as efforts to eliminate competitors (e.g., certain predators, herbivores, and insects) viewed as directly or indirectly reducing ranch profits. In some cases, these wildlife may have played an important role in keeping WPs in check, and

their systematic elimination may have opened the door for WPs to increase in abundance (e.g., prairie dogs; Weltzin et al. 1997).

The 1940s and 1950s

During the post–World War II era, heavy equipment and chemicals were readily available and were used on a broad scale. Our understanding of ecosystem processes and ecosystem goods and services was in its infancy during this period, and few environmental regulations were in place. Applied research in range science focused on the development and application of herbicides and mechanical techniques (Scifres 1980; Bovey 2001), often with the goal of eradicating shrubs. Brush management during this period was typically applied indiscriminately.

The 1960s and 1970s

Efforts aimed at widespread eradication in the 1940s and 1950s gave way to efforts aimed at selective control and containment in the 1960s and 1970s. By this time, it was clear that there were no "silver bullets" for brush management. Unlike many herbicide products available today that provide a long treatment life (15–50 yr; McDaniel et al. 2005; Perkins et al. 2006; Combs 2007), treatments in the past were relatively short lived (Jacoby et al. 1990a). Following chemical spraying, shrub cover often returned to pretreatment levels (or higher) within 5–15 yr. The necessity of retreating landscapes at relatively high frequencies made brush management nonsustainable and difficult to justify when cost often exceeded revenues generated from subsequent livestock production.

Basic and applied research from the 1940s to the 1970s led to the realization that brush management practices:

- were treating symptoms (the shrubs) rather than root causes of land cover change (e.g., disruption of historic grazing and fire regimes);
- must be conducted in concert with progressive livestock grazing management;
- when applied in an indiscriminant manner without careful planning, can
  - be detrimental to wildlife populations,
  - lead to homogenization and loss of biological diversity,
  - increase risks for catastrophic soil erosion and weed invasion, and
  - be too costly for a ranching enterprise, and
- can be short lived, with shrubs reestablishing dominance in 5–10 yr.

Collectively, these realizations led to the development of integrated brush management systems (IBMS) in the 1980s (Scifres et al. 1985; Brock 1986; Hamilton et al. 2004).

1980s to Present: IBMS

IBMS are long-term planning processes that move away from a purely livestock production perspective and toward management of rangelands for multiple uses and values. The IBMS planning process begins by identifying management goals and objectives for a specific site and the surrounding management unit. These might include increasing forage production; maintaining or promoting suitable wildlife habitat; augmenting stream flow or groundwater recharge; controlling pests, pathogens, or invasive species; maintaining scenic value; reducing wildfire risk; or preserving grassland and savanna ecosystems. Specific objectives are refined on a comprehensive inventory of ecosystem components (plants, animals, and soils), projecting the responses of those components to brush treatment alternatives, and considering the effects of treatment alternatives on management goals on other sites (Hanselka et al. 1996). Brush management techniques (herbicidal, mechanical, and prescribed burning) differ with respect to environmental impacts, implementation costs, efficacies, and treatment longevities. Thus, the IBMS approach advocates consideration of the type and timing of a given brush management technology and makes explicit allowances for consideration of the type and timing of follow-up treatments. This, in turn, requires knowledge of how woody and herbaceous plants will respond and how climate, soils, topography, and livestock and wildlife management might mediate plant responses. The IBMS approach is therefore inherently interdisciplinary and dependent on the active collaboration of a diverse group of management professionals.

Examples of the IBMS approach abound (Teague et al. 1997; Grant et al. 1999; Paynter

"IBMS are long-term planning processes that move away from a purely livestock production perspective and toward management of rangelands for multiple uses and values."

> The type, timing, and sequencing of brush management are the keys to long-term success"

and Flanagan 2004; Ansley and Castellano 2006; Ansley and Wiedemann 2008). Nearly every State Cooperative Extension Service in the western region offers a long list of guides and publications that address the IBMS approach and give specific recommendations for management of important WPs. There are expert system tools (e.g., PESTMAN 2009) available to assist land managers in selecting appropriate brush management practices and techniques (Hanselka et al. 1996). There are also special adaptations of the IBMS approach for small landholdings (McGinty and Ueckert 2001). NRCS guidelines should promote the IBMS approach and explicitly utilize it in the conservation planning process.

BRUSH MANAGEMENT: CURRENT PERSPECTIVES

> Brush management efforts must be viewed as an integral part of the overall system for wise, efficient use and conservation of grasslands. Available brush management and conservation methods are complex tools, the effectiveness of which depends primarily upon the resource manager's understanding of their proper application approached with consideration of all potential uses of rangeland. (Scifres et al. 1983, p. 11)

This statement, made over 25 yr ago, is still relevant today. This rationale is echoed in recent textbooks addressing brush management and ecosystem restoration (Valentine 1989; Whisenant 1999; Bovey 2001; Hamilton et al. 2004). The NRCS (2006) currently recognizes six reasons for undertaking brush management:

- Restore natural plant community balance
- Create the desired plant community
- Restore desired vegetative cover to protect soils, control erosion, reduce sediment, improve water quality, and enhance stream flow
- Maintain or enhance wildlife habitat, including that associated with threatened and endangered species
- Improve forage accessibility, quality, and quantity for livestock
- Protect life and property from wildfire hazards

A Texas survey found that the two primary goals of landowners investing in brush management were to increase forage production and to conserve water (Kreuter et al. 2005). Other reasons included improvement of aesthetic values, benefit the next generation, improve wildlife habitat, and improve real estate value.

Short- and Long-Range Planning Is Essential

A well-thought-out, comprehensive resource management plan reflecting short-, medium-, and long-term goals and objectives should be in place before attempting brush management.

In areas where shrubs are dense, herbicides or mechanical treatments, such as this crawler tractor equipped with push blades and discs, may be initially required to open up areas to promote herbaceous production and enable the subsequent use of prescribed fire as a management tool. (Photo: Tim Fulbright)

Evaluating multiple scenarios is useful for explicitly assessing the advantages and disadvantages of several alternatives. Given that the type, timing, and sequencing of brush management are the keys to long-term success, management plans should identify long-term objectives and work to ensure that resources and commitments are in place at the right time. Programs should include explicit information on what constitutes success and, if feasible, should address alternatives to primary objectives. Objectives will vary with the specific needs of the landowner, the community, and context of the action. Well-defined short- and long-term objectives are critical to determining when, where, why, how, and under what conditions brush management should be undertaken.

Considerable momentum is now building for landscape-scale IBMS projects. Such projects usually require close consultation, coordination, and cooperation among multiagencies and associated stakeholders. Managing rangeland brush and weeds requires adapting management methods to match species physiological and morphological traits and ecological site conditions (Fig. 7). The decision-making process associated with landscape-scale IBMS projects can be restricted by numerous factors, including equipment availability, financial constraints, available manpower, time needed to complete the task, environmental regulations, and agency mission and policy.

The Need for Monitoring

Monitoring is intrinsically linked to the IBMS process (i.e., treatment monitoring, control, revegetation (natural or planted), and pre- and posttreatment monitoring; Scifres et al. 1983; Scifres 1987; DiTomaso 2000). Assessing baseline (pretreatment) conditions is essential to determine the effects of brush management efforts. Monitoring of weather and seasonal growth of the plants targeted for manipulation should begin well in advance (3–6 mo) of planned treatment. Posttreatment monitoring should be conducted at least annually to evaluate responses.

Field variables to be measured for baseline inventories and posttreatment monitoring should be selected on the basis of the objectives of the brush management project. The following represent initial field variables that should be assessed in baseline inventories and to evaluate posttreatment responses:

- Shrub age (or size as a rough proxy), class distribution (baseline only), height, and stem density and diameter
- Plant composition and frequency of occurrence
- Ground cover of grass, forb (total and by species), litter, and bare ground
- Biomass (seasonal or peak live standing crop; preferably by species)
- Specific metrics will depend on the goals and objectives of the brush management project.

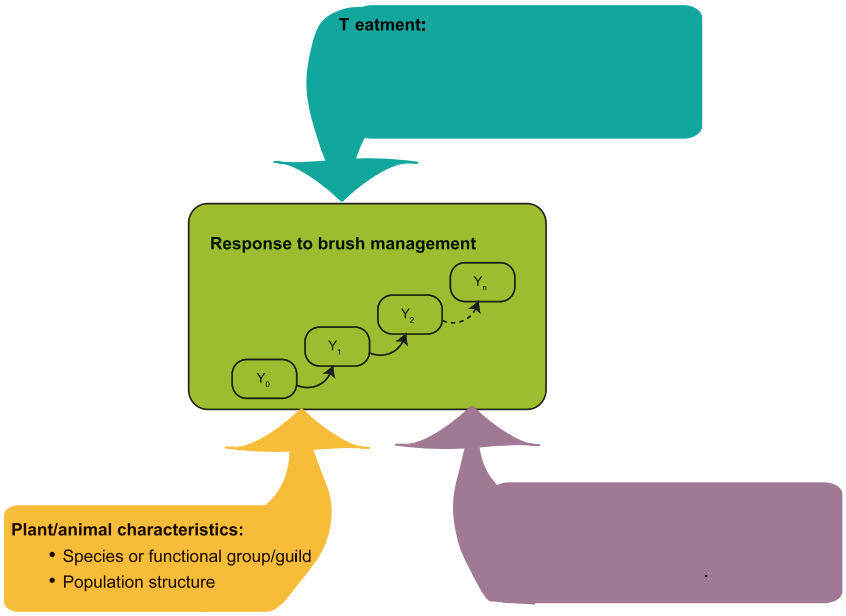


FIGURE 7. Ecosystem response to brush management varies with time since treatment (Y_1, Y_2, etc.) and is determined by a variety of interacting factors.

Treatment Options

Understanding the ecology of WPs and herbaceous plants and how they interact with each other on a particular site is crucial in determining the IBMS strategy. Relevant stand characteristics include plant community composition, plant phenology, plant density, plant size (stem diameter; canopy area, volume, and height), canopy architecture, and patterns of biomass allocation to leaves and stems aboveground and roots belowground. Each of these can influence the effectiveness and longevity of a given brush management practice. Realization of brush management objectives often benefits from spatially explicit prescriptions that take into account

TABLE 5. Factors influencing the effectiveness and longevity of a given brush management practice. Cookbook or "one-size-fits-all" approaches for brush management seldom succeed. A given project should be tailored to individual goals, objectives, and circumstances, and these, in turn, will be mediated by the items shown in the table.

• Site accessibility and terrain: If the site is difficult to reach or traverse (e.g., sandy soils, uneven topography), less labor-intensive methods (e.g., aerial spraying) might be more effective.
• Stand characteristics: Extent, age, biomass, and plant density will be factors in selecting the most cost-effective methods.
• Proximity to endangered species: The presence of a federally listed species or environmentally sensitive sites may preclude some types of management methods or limit the season of application.
• Presence of desirable plant species or other important resources (e.g., archaeological sites): Locally targeted methods (e.g., individual plant treatments) may be warranted to protect other resources in the area.
• Extent of area to be treated: Suitability and efficiency of a given treatment method may vary by the size of the area targeted for treatment.

the topoedaphic heterogeneity of landscapes (i.e., uplands, side slopes, riparian, valley bottomland, etc.; Taylor and McDaniel 2004; Table 5).

BRUSH MANAGEMENT AS A CONSERVATION TOOL: A CRITICAL ASSESSMENT

The projected effects of brush management on ecosystems and ecosystem processes boils down to its effects on: 1) herbaceous vegetation and native plant communities; 2) livestock; 3) watershed properties related to erosion, soil condition, water quality, and water quantity; 4) wildlife; 5) air quality; and 6) human dimension considerations. The following sections summarize the scientific literature addressing the first five of these and the question, Are the outcomes expected from brush management being realized? For treatment of economics and the human dimensions area, see Tanaka et al. (this volume).

Our assessment began with a series of literature searches using the Web of Knowledge℠. Search strings included "brush management" and terms referring to specific brush management techniques. Search results were filtered to include only studies conducted in the United States on rangelands and only those quantifying responses to brush management. Studies quantifying herbaceous responses dominate the brush management literature in the United States and comprised 48.7% of those in the sample (Fig. 8). Treatment efficacy and shrub regeneration studies accounted for another 28.7%. Studies documenting water (4.1%) and soil (2.0%) responses were the least common.

Herbaceous Vegetation and Native Communities

The general expectations associated with brush management are that it will mitigate soil erosion, improve soil condition, enhance water quantity and quality (via improvement in infiltration and reductions in runoff, which interact to reduce sedimentation), and improve livestock production. Each of these expectations is based on the assumption that herbaceous ground cover will increase following brush management. How good is this assumption?

Herbaceous Response. The majority (>80%) of studies in our literature sampling reported positive herbaceous responses following brush management (Appendix I). Herbaceous plant growth increases an average of 3- to 5-fold for brush management conducted on productive range sites, including sites with Wyoming big sagebrush (*Artemisia tridentata*; Hyder and Sneva 1956; McDaniel et al. 2005) and broom snakeweed (*Gutierrezia sarothrae*; McDaniel et al. 1993). Management of other woody species, including mesquite (*Prosopis glandulosa*) and creosote bush, can result in substantial forage increases on productive sites with adequate rainfall (Ethridge et al. 1984; Perkins et al. 2006; Combs 2007). In semidesert grasslands at the Santa Rita Experimental Range in Arizona, herbage yields following velvet mesquite removal increased (Parker and Martin 1952; Paulsen 1975; Cable 1976) or remain unchanged in zones <1100 m in elevation and when velvet mesquite cover was <25% (McClaran and Angell 2006). These patterns are consistent with field studies in southern New Mexico (Warren et al. 1996; Drewa and Havstad 2001).

No consistent relationship between posttreatment changes in herbaceous production and annual rainfall were found;

however, a survey of data across a range of management contexts suggests an upper limit for the herbaceous production responses that might be expected for a given rainfall zone (Fig. 9A). Herbaceous response corrected for annual rainfall varies with time since brush treatment (Fig. 9B). The median first-year response is 0 and highly variable, with half the treated sites responding positively and half negatively. By year 2, the median response is slightly positive but also highly variable. After year 2, the response becomes more consistent and peaks in year 5. The response then drops off in years 7 and 8, being slightly but consistently positive.

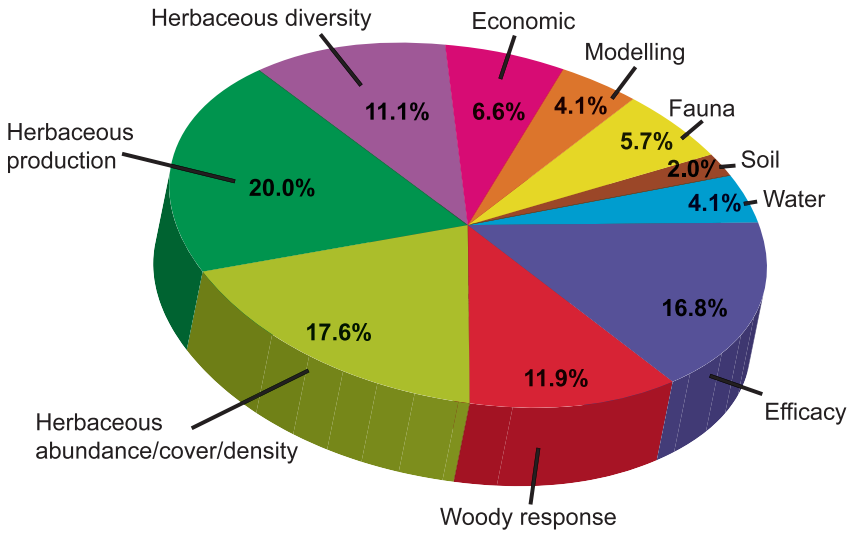
The longevity of brush management treatments varies widely by type of treatment applied, shrub species, effectiveness of the initial treatment, composition of the herbaceous vegetation, and soil properties (Fig. 10). Variations in the Figure 10 conceptual model have been illustrated for velvet mesquite (Cable 1976), honey mesquite (Heitschmidt et al. 1986; Ansley et al. 2004b), big sagebrush (McDaniel et al. 2005), and creosote bush (Gibbens et al. 1987; Morton and Melgoza 1991; Perkins et al. 2006). The change in foliage cover and herbaceous response to brush management ranges from 5 to 20 yr for velvet mesquite (Cable 1976), from 10 to 25 yr for honey mesquite (Dahl et al. 1978; Jacoby et al. 1991, 1990a, 1990b; Combs 2007), >25 yr for sagebrush (McDaniel et al. 2005), and >40 yr for creosote bush (Perkins et al. 2006). The general curve shape of the overstory–understory relationship for these shrub species is similar, but average grass yield associated with overstory cover is scaled quite differently: from 2- to 3-fold greater for mesquite relative to big sagebrush and creosote bush, respectively. Mesquite management typically provides a greater forage response, but it is of shorter duration than for big sagebrush and creosote bush removal. Accordingly, timing of investments to re-treat communities dominated by these shrubs would be on the order of about 4–12 yr for mesquite, 20–30 yr for big sagebrush, and >30 yr for creosote bush (Torell and McDaniel 1986; Torell et al. 2005a).

Although studies have or currently are being conducted across different ecological sites in the western United States, adequate data to statistically estimate the relationships in Figure 10 as a function of rainfall, soils, site productivity, and so on are not generally available. Instead, qualitative assessments by experienced range scientists and economists are currently the norm for projecting forage response to brush management (Fig. 11).

Soil Condition. Some of the projected effects of brush management on soil are associated with the assumption that soil organic matter depletion will occur with shrub encroachment. However, as reviewed earlier (see the section "Soil Condition and Erosion"), this is not a robust assumption. It may be true in certain cases, most likely those in arid areas where disruption of grass cover by grazing has accelerated wind and water erosion. But even in those instances, soil resources may undergo not a net change in abundance but, rather, a change from a homogenous to a heterogeneous distribution wherein they are concentrated within shrub islands (Schlesinger and Pilmanis 1998). Thus, were it not for shrubs, soil resources may have been lost from the site because of grazing rather than being spatially rearranged. Brush management, by reducing shrub cover on fertile shrub islands, may put

> "Robust generalizations regarding brush management effects on soil carbon are currently not possible."

FIGURE 8. Proportion of brush management studies quantifying various categories of treatment effects. Published papers resulting from Web of Knowledge search strings that included "brush management" and terms referring to specific brush management techniques were distilled to a database of 333 articles that reported quantitative responses. These were then classified into the categories shown. Articles reporting data for multiple metrics were tallied in multiple categories. Thus, the graph reflects the information reported in the literature but not on a per-paper basis.



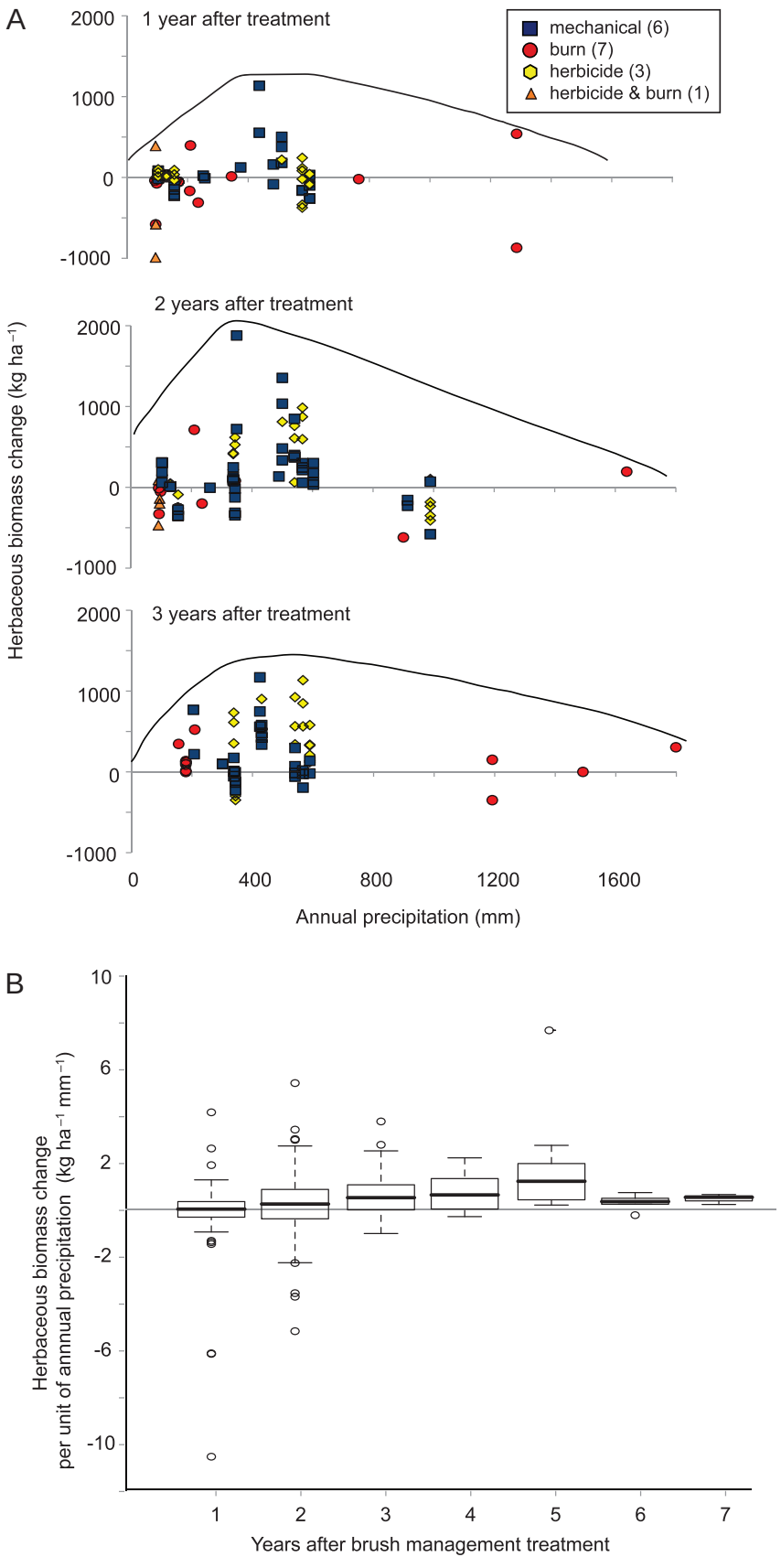


FIGURE 9. (**A**) Changes in herbaceous biomass production (kg·ha^{-1}) 1, 2, and 3 yr after brush management as a function of current years' annual precipitation (PPT, mm). Multiple observations for a given PPT value reflect multiple sites or different brush management applications. PPT was determined from nearby weather stations if not reported. The number of studies pertaining to a given brush treatment are listed parenthetically in the key. (**B**) Change in herbaceous biomass per millimeter of annual precipitation received after brush management. Responses are from 13 studies representing brush management with fire, herbicides, and mechanical treatments. Tukey box plots show inner-quartile range (IQR; rectangle) and the median (bold line). Whiskers indicate the maximum and minimum values or the values within 1.5*IQR of the third and first quartiles, respectively. Values beyond 1.5*IQR of the first and third quartiles are considered statistical outliers and are indicated with open circles. *N* = 13, 13, 11, 8, 5, 3, and 2 for years 1 through 7, respectively. The number of studies pertaining to a given brush treatment are listed parenthetically in the figure legend. Citations used to generate the data points are given in Appendix III.

these sites at risk for net losses of soil nutrients unless ground cover is quickly established. Alternatively, nutrient losses from shrub islands following brush management may help reinstate the homogeneous distribution of resources by disrupting the processes that lead to the concentration of nutrients in and around shrub canopies (e.g., Davies et al. 2009a; Ravi et al. 2009). In the latter scenario, the likelihood of getting grasses reestablished within intershrub zones may improve. Site-specific factors may dictate which of these competing scenarios is most likely on a given landscape.

WP encroachment can have a moderate to strong positive impact on SOC and N pools on many sites (Fig. 4). This shrub-induced improvement in SOC and N may be an important factor underlying the extent to which herbaceous vegetation production increases following brush management (Fig. 9). The degree to which shrubs might increase soil resources beneath their canopies is a function of how long the shrubs have occupied the site (Wheeler et al. 2007; Throop and Archer 2008). Thus, stand age at the time of brush management will have an important bearing on soil conditions. Removal of individual shrubs causes depletion

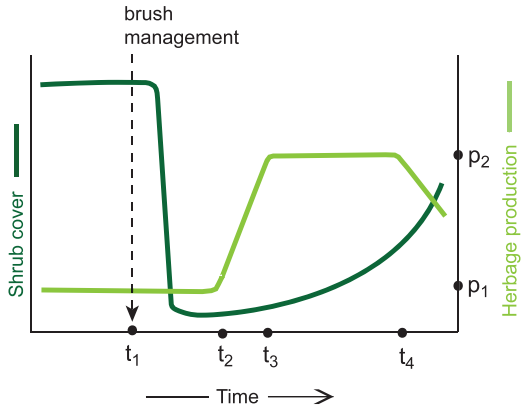


FIGURE 10. Generalized conceptual model of herbaceous response to brush management. The lag time in response (t_1–t_2), the magnitude of (p_1–p_2) and time (t_1–t_3) to peak herbaceous response, the duration of peak elevated production response (t_3–t_4), and the time frame over which herbaceous productions decline as shrubs reestablish (t_4 onward) vary with numerous factors. Knowledge of the relationships depicted in this conceptual model for a given ecological site will help determine the type, timing, and appropriate sequencing of brush management practices in an Integrated Brush Management Systems (IBMS) approach.

of the associated resource pool and the availability of nutrients over the 10–15 yr following treatment, the extent depending on whether shrubs regenerate (Klemmedson and Tiedemann 1986; Tiedemann and Klemmedson 1986, 2004). Losses of SOC and N accumulating in soils associated with mature shrubs killed by herbicide ranged from 67% to 106% at 0–5-cm soil depths and from 78% to 93% at 5–10-cm soil depths over a 40-yr period (McClaran et al. 2008). Data from these individual plant perspectives suggest that brush management will cause a decline rather than an increase in SOC and N pools in hot, semidesert rangelands but that shrub regeneration can arrest or reverse such declines (Hughes et al. 2006). These findings contrast with those of Teague et al. (1999), who compared SOC and N on sites 4–22 yr after root plowing against untreated controls in the southern Great Plains to test the hypothesis that removal of honey mesquite would result in steady decline in SOC because of a loss of mesquite inputs and reductions in shading (and therefore higher soil temperatures and higher oxidation rates). Overall, they found no significant differences between treated and control sites. Similarly, Hughes et al. (2006) found that while aboveground C and N pools increased markedly with mesquite stand development following brush management (more so on sandy sites than shallow, clayey sites), near-surface SOC and N pools were unaffected. Thus, as with WP encroachment (Fig. 4), robust generalizations regarding brush management effects on soil condition are currently not possible.

> "Timing of investments to re-treat communities is about 4–12 yr for mesquite, 20–30 yr for big sagebrush, and >30 yr for creosote bush"

FIGURE 11. Projected changes in livestock stocking rates for aerial spraying and mechanical plus grass seeding treatment of a *Prosopis*–mixed brush system in southern Texas relative to untreated controls (J. Conner et al., unpublished data, 1998).

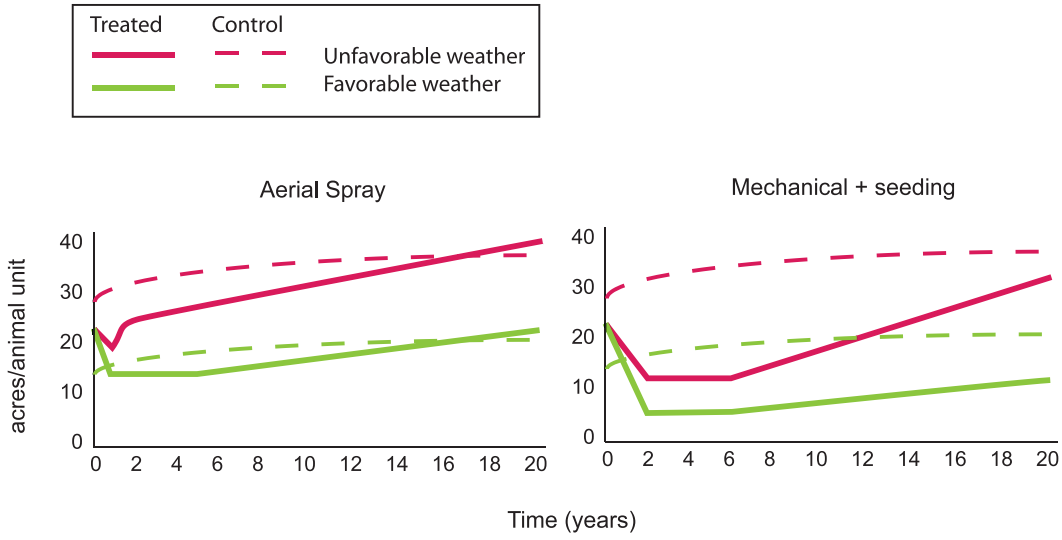


FIGURE 12. (**A**) Mojave Desert scrub near Las Vegas, Nevada, USA (foreground) and area invaded by the nonnative annual grass red brome (*Bromus rubra*; background) following a fire that carried from desert floor upslope into pinyon–juniper woodlands. This exotic grass has instigated a positive disturbance (fire) feedback that reduces ecosystem carbon storage, threatens the biodiversity, and constitutes a new ignition source for fire in upper-elevation woodlands and forests. Photo: T. E. Huxman. (**B**) Nonnative annual grass (medusahead, *Taeniatherum caput-medusae*) invaded area (foreground) transitioning into a Wyoming big sagebrush (*Artemisia tridentata wyomingensis*) plant community in southeastern Oregon. The presence of this exotic annual grass increases the probability that the sagebrush plant community above it will burn. (Photo: K. W. Davies)

Non-native species invading or purposely seeded following brush management (see Fig. 12 and the section "Biodiversity and Nonnative Species") may significantly reduce ecosystem C accumulating with WP encroachment. Indeed, estimates of aboveground C loss with conversion of Great Basin shrublands and woodlands to annual grasslands are on the order of 8 Tg C, with estimates of 50 Tg C release to the atmosphere over the next several decades (Bradley et al. 2006). In cold desert sagebrush steppe ecosystems, this level of C release with annual grass invasion could completely offset any increases in C with woody encroachment that has occurred over the past century. However, the story may be quite different in southwestern rangelands where highly productive, deeply rooted perennial grasses introduced from Africa are expanding and sequester substantially more C than annual grasses (e.g., Williams and Baruch 2000; Franklin et al. 2006).

Specific brush management techniques will likely differ in their impact on litter decomposition, depending on the type of disturbance they cause, treatment efficacy, and the extent to which they co-occur with other land use practices, such as livestock grazing. Brush management treatments that minimally disturb soils (e.g., herbicide applications and prescribed burning) may be most advisable for managers wishing to minimize short-term SOC losses. In contrast, brush management techniques that cause extensive disturbance to the soil surface, such as chaining, root plowing, and grubbing, may increase decomposition rates due to surface soil disturbances. These practices likely superimpose a variety of new short- and long-term direct and indirect effects on decomposition processes via their dramatic alteration of surface roughness, water infiltration and runoff, vegetation cover, and ANPP and by initiating large, synchronous inputs of leaf, stem, and coarse woody debris onto the soil surface with widely varying degrees of contact and incorporation into the soil. Such treatments ostensibly increase exposure to direct sunlight and UV radiation and may promote soil movement via wind

and water, particularly during the immediate posttreatment period when vegetation is reestablishing.

Vegetation responses to brush management have been widely described, but very little is known of its effects on soils and nutrient cycling (Fig. 8). There have been few attempts to model brush management effects on ecosystems (but see Carlson and Thurow 1996; Grant et al. 1999), and the future development of such models would likely benefit from field studies elucidating how various brush management practices might impact C and N cycling. Lessons learned from studies of temperate forest clear-cutting and tropical deforestation would be instructive starting points, but it is likely that the shrublands, savannas and woodlands of drylands would have novel behaviors. For example, predicting brush management effects on litter decomposition in semidesert grasslands will require information on shrub–grass interactions and herbaceous biomass influences on soil movement at a decadal time scale (Throop and Archer 2007).

Biodiversity Response. Biodiversity responses can be assessed at the species (e.g., genetic variation in populations), the organismal (species richness), the structural (vegetation strata and physiognomy), and the functional (plant functional groups and animal guilds) levels. Studies at the organismal level are typically restricted to a select class of organisms (e.g., perennial herbaceous plants or small mammals) without regard for other classes (annual plants, shrubs, reptiles, avifauna, large mammals, microbes, etc.). To further complicate things, diversity varies with scale (e.g., alpha, beta, and gamma diversity) and topoedaphic heterogeneity. Objectives aimed at preserving, restoring, and monitoring biodiversity should thus be phrased to specifically articulate the facets of biological diversity being addressed.

From a Web of Knowledge search generating 333 studies quantifying responses to brush management (Fig. 8), 39 articles reporting herbaceous plant diversity emerged; of these, 13 were conducted on rangelands and were amenable to comparative analysis. From the 90 data points emerging from these studies, it appears that brush management treatments typically have neutral (30% of data points exhibiting <10% change) to positive (60% of data points exhibiting >10% increase) effects on grass/forb diversity (Fig. 13). Cases where brush management had negative effects on herbaceous diversity (10% of data points exhibiting >10% decline) were typically associated with herbicide treatments, ostensibly reflecting adverse impacts on forbs. The few long-term data available suggest that posttreatment stimulation of herbaceous diversity is relatively short lived (<15 yr).

In the subtropical southern Great Plains characterized by a diverse flora of encroaching WPs, WP communities developing after brush management have lower shrub diversity and higher densities of less desirable browse species than the previously existing community (Fulbright and Beasom 1987; Ruthven et al. 1993). In systems where shrubs aggressively regenerate vegetatively, use of low-intensity fire and herbicides can promote a savanna physiognomy (e.g., Ansley et al. 1997, 2003) and ostensibly promote diversity.

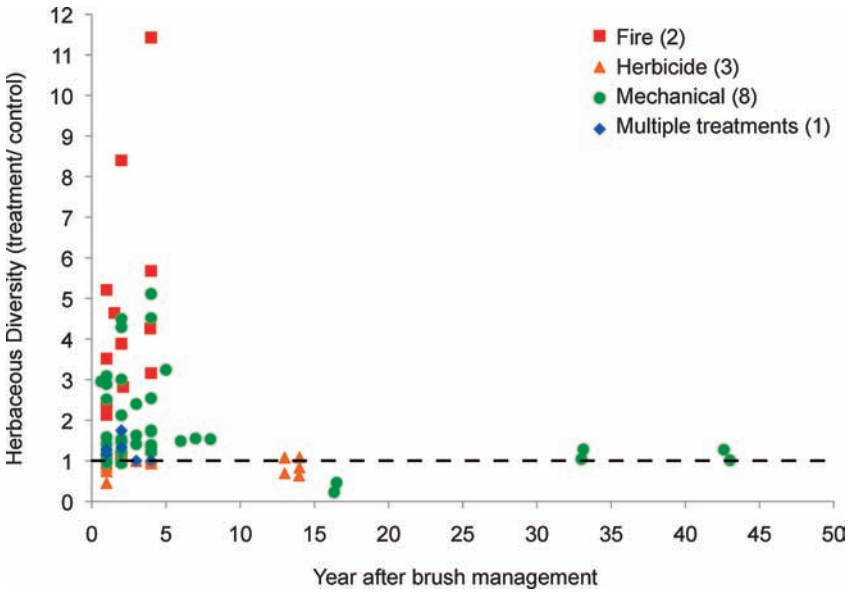


FIGURE 13. Changes in herbaceous vegetation diversity following brush management. Results (90 data points) show ratios of values (richness and various indices) reported for treated and control areas. Ratios near 1.0 indicate that diversity metrics on treated and control areas were similar, values <1.0 indicate that brush management decreased diversity, and values >1.0 indicate increases in diversity. The number of studies pertaining to a given type of brush treatment are listed parenthetically in the key. See Appendix III for citations.

“Brush management has the potential to create conditions favorable for the establishment and growth of weeds and invasive nonnative species.”

Faunal diversity response to brush management varies with the organisms of interest (see the section “Wildlife”). For example, although Jones et al. (2000) reported that relative total abundance and species richness of herpetofauna was similar among a variety of treatments, amphibians were most abundant in untreated and herbicide-only sites, lizards were most abundant on untreated sites, and snakes were most abundant on sites receiving herbicide and fire. Rodent and avian relative frequency, richness, and diversity have been observed to be unaffected by brush management (Nolte and Fulbright 1997; Peterson 1997).

The biodiversity response to brush management may be strongly influenced by the pattern of treatment application (see Bestelmeyer et al. this volume). A “wall-to-wall” application may yield one result, whereas applying a treatment or combination of treatments in “strips” may have a quite different outcome by creating more habitat edge and creating patches of grassland habitat interspersed with shrubland habitat (e.g., Scifres et al. 1988). For example, diversity of native perennial grasses may be promoted by a mixture of open areas interspersed with cover of mature shrubs (Tiedemann and Klemmedson 2004). Effects of brush management on biological diversity are poorly understood and need to be investigated at larger scales across longer time periods.

Biodiversity and Nonnative Species. Brush management has the potential to create conditions favorable for the establishment and growth of weeds and invasive nonnative species (Young et al. 1985; Belsky 1996; Bates et al. 2007) that can have adverse affects on biodiversity. Brush management is therefore often conducted in conjunction with seeding operations intended to accelerate establishment of ground cover and a forage base (see Hardegree et al. this volume). In many cases, the grasses used for seeding are nonnative perennials (Cox and Ruyle 1986; Ibarra-Flores et al. 1995; Martin et al. 1995; Christian and Wilson 1999). Seeds from such species may be more readily available, and their establishment success rates may be higher than that of natives (Eiswerth et al. 2009). When seeding of nonnative grasses is successful, the result is often a persistent, long-lived near-monoculture of nonnative vegetation. While this may be valued for livestock production and ground cover and may make the site more resistant to invasion by undesirable exotic annual grasses (Davies et al. 2010) by virtue of their superior competitive ability (Eissenstat and Caldwell 1987), these plants may represent threats to the biodiversity of native plants and animals (McClaran and Anable 1992; Williams and Baruch 2000; Schussman et al. 2006). Their unintended spread into areas beyond where they were planted may make it difficult to achieve conservation goals on other lands. Thus, there are clear trade-offs that should be explicitly considered and evaluated.

A Tool to Promote Landscape Heterogeneity and Biodiversity? Disturbances associated with fire and herbivory (granivory, grazing, browsing, burrowing, trampling, and dung/urine deposition) interact with climate variability and extremes to generate patchiness across the landscape and contribute to the maintenance or enhancement of biological diversity. It is now recognized that such disturbances should be explicitly included in ecosystem management and conservation plans (Pickett et al. 1997; Fuhlendorf and Engle 2001).

In the IBMS approach, brush management techniques can be targeted for certain portions of a landscape and distributed across landscapes in both time and space such that mosaics of vegetation structures, patch sizes, shapes, and age states are created. This, in turn, would promote the co-occurrence of suites of insect, reptile, mammalian, and avian species with diverse habitat requirements (Jones et al. 2000). The logistics of planning and applying spatially heterogeneous brush management practices at appropriate scales is facilitated by advances in geomatics (e.g., global positioning satellites, geographic information systems, and remote sensing imagery) and landscape ecology that allow habitat and population data to be readily linked over large areas. Thus, a low-diversity shrubland or woodland developing on a grassland site can be transformed to a diverse patchwork of grassland–savanna–shrubland communities using a spatial placement of landscape treatments that promotes biological diversity at multiple scales (Scifres et al. 1988; Fulbright 1996).

Rocky Mountain elk in a sagebrush community where encroaching western juniper trees have been cut to preserve habitat for sagebrush associated wildlife. (Photo: K. W. Davies)

Livestock Response

Livestock grazing contributes significantly to the economy and social fabric of most rural communities. Brush management is a tool used to restore native ecosystems that have the capacity to provide a steady source of forage for livestock while facilitating other uses and resource values (NRCS 2006; US Department of the Interior, Bureau of Land Management 2007). The decision of whether to apply brush management for the betterment of domestic livestock is influenced by numerous factors, including the extent to which declines in carrying capacity (Olson 1999), animal performance (Ralphs et al. 2000), animal loss from poisoning (Williams 1978; Panter et al. 2007), animal handling (Hanselka and Falconer 1994), and animal health (Teel et al. 1998) will be impacted. Even when grazing has contributed to shrub increases, simply removing livestock or reducing their numbers is unlikely to remedy a brush encroachment problem (Browning et al. 2008). Passive treatments may help, but in many situations aggressive intervention is necessary (Olson 1999). Livestock can be used as part of the vegetation treatment program, especially when goats and sheep are used to apply browsing pressure on

unwanted shrubs and weeds (Riggs and Urness 1989; Frost and Launchbaugh 2003).

Studies quantifying forage response to reductions in brush cover are relatively numerous (Fig. 8), but few have quantified direct commodity (livestock) benefits. Potential changes in livestock carrying capacity for contrasting brush management × precipitation scenarios are illustrated in Figure 11. These projections illustrate that a range in livestock returns should be anticipated because of differences in forage response during favorable, normal, and unfavorable rainfall conditions that may occur over a 20-yr horizon. In this example, using cattle prices equal to the average of the past 20 yr, current operating costs, and current costs of brush management practices, the returns on aerial spraying and mechanical practices may be relatively high when environmental conditions support high levels of herbaceous production. However, returns are greatly reduced when conditions for plant growth are poor. Subjective projections such as these are based on the best available information, and actual results are known to vary widely, depending on the specific situation. As research continues, more accurate and reliable projections can be developed.

Increases in available forage following brush management do not necessarily warrant an increase in livestock numbers. In some cases, justification for brush management may be to maintain stocking rates nearer its true capacity (i.e., recognition that current stocking rates cannot be sustained). For example, big sagebrush management on public land helps avoid potential conflict and lawsuits with grazing permittees and environmentalists because positive steps are taken to reduce grazing pressure without forcing major herd reductions (Torell et al. 2005a). Similarly, forestalling the need for controversial grazing reductions was a primary benefit of the 11-yr (1962–1972) Vale Rangeland Rehabilitation Program initiated in eastern Oregon (Bartlett et al. 1988). In the case of big sagebrush, brush management is not always acceptable because of its adverse impact on habitat for sagebrush-obligate wildlife species (Rhodes et al. 2010).

Reductions in brush and weeds potentially benefit livestock operators by increasing grazable land area (McDaniel et al. 1978). However, returns based solely on gains in animal performance are not always economically justified, especially when public assistance is not available (McBryde et al. 1984; Torell et al. 2005b). Lee et al. (2001) found that costs for brush management projects in the Edwards Plateau area of Texas exceeded livestock returns by 7–31%. Similarly, Torell et al. (2005b) found that a 30% cost-share agreement was required to justify big sagebrush management in New Mexico when the added forage from the brush management practice was valued at an intermediate level of $7/AUM (in 2003 dollars). A range improvement practice that increases forage during critical or limiting seasons can be economically feasible (Evans and Workman 1994).

While other resources (soil, water, wildlife, etc.) may benefit from IBMS, the economics of brush management practices continue to be evaluated on the basis of the amount of forage and meat products gained by implementing the practice (Tanaka and Workman 1988; Watts and Wamboldt 1996; Lee et al. 2001). The economic component of the holistic decision support system PESTMAN (2009) is driven by the anticipated forage response to brush management. Yet, as noted over 30

At advanced stages of shrub encroachment, brush management can improve biological diversity while potentially benefitting livestock production by increasing grazable land area. (Photo: K. W. Davies)

FIGURE 14. Overland flow is an important mechanism of runoff generation for many semiarid landscapes. For woodlands such as the pinon–juniper stand in the Jemez Mountains, New Mexico, pictured here, much of the runoff and erosion is generated from the intercanopy spaces. (Photo: Bradford Wilcox)

yr ago by Smith and Martin (1972), most range improvements show a negative benefit/cost ratio (costs exceed benefits) when based only on the value of the added forage for livestock production. This is a consistent and continuing conclusion that increased returns from improved animal performance and production are often too low for brush management to be economically justified (McBryde et al. 1984; Lee et al. 2001; Torell et al. 2005a). Landowners recognize this, and many if not most recognize other benefits to conducting brush management beyond livestock production. Additionally, most landowners conducting a brush management project do so under cost-share arrangements with state and federal agencies. When the value of ecosystem goods and services beyond those associated with livestock production are taken into account, a more favorable picture of brush management begins to emerge (see Tanaka et al. this volume).

Watershed Function

The NRCS makes a number of assumptions related to the hydrological consequences of brush removal. These assumptions fall into three broad categories: 1) horizontal fluxes—the removal of WPs will reduce overland flow (surface runoff) and erosion, primarily by improving infiltration rates and increasing ground cover; 2) vertical fluxes—the removal of WPs will reduce the evapotranspiration (ET) and thus increase groundwater recharge; and 3) landscape effects—as a result of assumptions 1 and 2, the removal of WPs will reduce gully erosion and increase stream flow. We review the validity of these assumptions below on the basis of relevant literature. Our review is organized by the primary geographic regions in the United States where information is available: the Southwest, the Northwest, and the southern Great Plains.

Horizontal Fluxes—Surface Runoff and Erosion. The expectation that surface runoff and erosion are higher from woodlands or shrublands than from grasslands is implicit in the assumption that reductions in WP cover will reduce overland flow and water erosion (Fig. 14). In some cases, this is true, but in many cases, it is not. It is most likely in the xeric climates that support creosote bush shrublands and piñon–juniper (*Pinus* spp.–*Juniperus* spp.) and juniper woodlands. The influence of woody species encroachment on surface runoff and erosion depends on the impacts of encroachment on herbaceous vegetation and subsequently bare ground. Surface runoff and erosion increase when WP

FIGURE 15. Reductions in herbaceous ground cover resulting from grazing and woody plant encroachment can increase connectivity between bare patches and lead to higher runoff and erosion (Davenport et al. 1998). (**A**) pinyon–juniper woodland in the Jemez Mountains, New Mexico, and (**B**) creosote bush shrubland in southern New Mexico. (Photo: Bradford Wilcox)

encroachment decreases herbaceous vegetation and increases bare ground; however, if WP encroachment does not decrease herbaceous vegetation and increase bare ground, then surface runoff and erosion would not increase. Brush management does not always reverse the impacts of WP encroachment on surface runoff and erosion. In some cases, depending on the woodland type and the method of shrub management, surface runoff and erosion may actually increase.

Southwest. There is clear evidence that as desert grasslands transition to creosote bush, juniper, or mesquite shrublands or woodlands, there is more bare ground and better-connected interspaces, resulting in lower net infiltration, more surface runoff, and higher erosion (Fig. 15) (Parsons et al. 1996; Schlesinger et al. 2000; Mueller et al. 2008; Wainwright et al. 2000). However, the reverse has not been demonstrated. In other words, brush management on creosote bush shrublands does not necessarily curtail surface runoff and erosion (Tromble et al. 1974; Tromble 1978, 1980; Wood et al. 1991).

There has been relatively little work evaluating the hydrological implications of managing mesquite in the Southwest. Long-term watershed studies at the Santa Rita Experimental Range suggest that runoff and sediment yields may decline with mesquite removal (Lane and Kidwell 2003). The results are, however, equivocal because no pretreatment monitoring took place.

Surface runoff is a relatively small portion of the water budget in piñon–juniper woodlands (Gifford 1975), primarily because of internal storage within the hillslopes (Reid et al. 1999; Wilcox et al. 2003a). Surface runoff is higher when snowmelt occurs (Baker 1984; Wilcox 1994). Infiltration rates are higher under tree canopies than in the interspaces spaces (Reid et al. 1999), even though the hydraulic conductivity of canopy and intercanopy soils is similar (Wilcox et al. 2003b), likely because of the buildup of duff under that canopy. In these woodlands, small-plot infiltration studies indicate that shrub management has little effect or even a negative effect on infiltration rates (Gifford et al. 1970; Blackburn and Skau 1974; Roundy et al. 1978). Runoff and erosion are the highest following chaining and windrowing. If debris is left in place, there is little difference in surface runoff between treated and untreated locations (Gifford 1973). Watershed-scale experiments in Arizona indicate little effect of brush management on surface runoff (Clary et al. 1974; Collings and Myrick 1966). Although runoff may be relatively small in general, it may be much higher on woodlands occurring on slopes >10% (Wilcox et al. 1996a, 1996b). In these landscapes, cutting trees and leaving the slash in place has proven to dramatically decrease erosion rates, primarily because of increases in herbaceous cover (Hastings et al. 2003).

Northwest. Major shrublands of the northwestern United States are those dominated by sagebrush or western juniper

(*Juniperus occidentalis*). Erosion on sagebrush rangelands is generally very low (Coppinger et al. 1991). Effects of brush management on horizontal water fluxes in sagebrush landscapes are equivocal. Increasing surface runoff and erosion has been documented in some cases, and declines have been observed in others (Blackburn 1983; Brown et al. 1985). Mechanical treatments that disturb soil can increase runoff and erosion. For example, plowing reduced infiltration and increased runoff and erosion up to 12 yr (Gifford 1982). Soil erodibility was initially ~2-fold greater in burned compared to unburned sagebrush communities, but was comparable 1 yr postfire, and infiltration and runoff was comparable on burned and unburned hillslopes in the first and second years postfire (Pierson et al. 2001). However, in another study, burning had little effect on runoff but resulted in a large increase in erosion (Pierson et al. 2008). Balliette et al. (1986) found little change in infiltration, runoff, or erosion following herbicide treatment. In contrast, Blackburn and Skau (1974) found that plowing and reseeding of big sagebrush increased infiltration rates and lowered surface runoff. Effects can also vary with season. At the small catchment scale (2–4 ha), summer runoff and erosion declined by 75% and 80%, respectively, following conversion of sagebrush to introduced grasses, whereas snowmelt runoff increased 12% (Lusby 1979). The contrasting results from this population of studies likely reflect differences in responses of the herbaceous understory to sagebrush clearing and differences in disturbance impacts associated with various brush management techniques.

Western juniper has been aggressively encroaching into sagebrush communities across the intermountain West (Miller et al. 2005). Subsequent to its establishment, western juniper excludes other vegetation and increases bare ground (Miller et al. 2000). Although Belsky (1996) found little compelling evidence that surface runoff and erosion were higher following western juniper encroachment, other work suggests that runoff and erosion can be significantly accelerated and that brush management can significantly mitigate these effects (Buckhouse and Mattison 1980; Gaither and Buckhouse 1983). Indeed, Pierson et al. (2007) found that 10 yr after juniper removal, treated hillslopes had significantly more vegetation cover, higher infiltration rates, and 15-fold less erosion than nontreated sites.

Southern Great Plains. In the southern Great Plains, the major shrublands of concern are those dominated by mesquite or juniper. Most of the research related to WPs, and water has been conducted in relation to Ashe juniper (*Juniperus ashei*) in the Edwards Plateau with some additional work completed on mesquite woodlands in the Rolling Plains and the South Texas Plains.

Extensive woodlands dominated by Ashe juniper occur on the Edwards Plateau of central Texas. As with other juniper woodland types, there is a widely held perception that encroachment by this WP has promoted surface runoff and erosion. However, there is little evidence in support of this assumption. Infiltration rates within this woodland type are relatively high, and erosion is low unless the area is heavily grazed (Hester et al. 1997; Wilcox et al. 2007, 2008b; Taucer et al. 2008).

For mesquite shrublands in the Rolling Plains of northern Texas, small-plot rainfall simulations indicate that shrub management may improve infiltration capacity and reduce erosion as a result of increased herbaceous cover (Bedunah 1982; Brock et al. 1982). Larger-scale plot and catchment studies, however, suggest the honey mesquite management would not significantly alter surface runoff and erosion (Carlson et al. 1990; Wilcox et al. 2006). Weltz and Blackburn (1995) reached a similar conclusion for mesquite–mixed shrub rangelands in the Rio Grande Plains.

Vertical Fluxes, ET, and Groundwater Recharge. WPs have the potential to alter the fluxes of water moving in a vertical direction, ET, and recharge by virtue of the fact that deep root systems allow WPs access to water not available to more shallow-rooted vegetation. The ability of WPs to access deep water is, however, modulated by soil depth, texture, and the underlying geological structure, the latter also being a key determinant of whether groundwater recharge events will affect stream base flow. In principle, in locations where WPs are accessing deeper water, there is the potential

> “There are few examples demonstrating that brush management enhances ground water recharge or streamflow.”

“Historical stream flow records in the Edwards Plateau indicate that base flows have actually increased substantially since 1960 in spite of the fact that WPs have increased markedly since that time.”

to use vegetation management to enhance recharge. In practice, however, there are few examples of where this has been demonstrated and then only at relatively small scales.

Southwest. There is little difference in ET between creosote bush shrublands and desert grasslands (Small and Kurc 2003; Kurc and Small 2004). Recent work suggests that removal of shrubs could increase groundwater recharge but not in amounts that would appreciably affect water supplies (Sandvig and Phillips 2006).

Recharge rates in most piñon–juniper woodlands are very small, and it is unlikely that brush management would lead to higher recharge (Newman et al. 1997; Sandvig and Phillips 2006). However, decreasing piñon–juniper cover by chaining increased soil moisture in the upper 60–90 cm of the soil profile, with only minor differences at greater depths (Gifford and Shaw 1973). We are not aware of any work comparing ET between piñon–juniper woodlands and comparable grassland areas.

Northwest. Removal of sagebrush can increase soil water content and presumably recharge (Sturges 1993; Seyfried and Wilcox 2006). Sturges (1993) suggested that reductions of sagebrush cover can increase water yield if sagebrush roots are not confined to the same volume of soil as grass roots. Along these lines, Darrouzet-Nardi et al. (2006) found that sagebrush in herbaceous meadows in the Sierra Nevada Mountains was in fact accessing deeper water than the herbaceous vegetation. Sagebrush management decreases water withdrawal from the upper 1 m of soil for 2 yr posttreatment (Sonder and Alley 1961; Cook and Lewis 1963; Tabler 1968; Shown et al. 1972; Sturges 1977). However, over longer periods of time, water depletion to 0.9-m soil depth can be greater where sagebrush is removed compared to where it is not because of increases in herbaceous vegetation production (Sturges 1993). The replacement of sagebrush by nonnative annual grasses and forbs (e.g., Fig. 12) can alter the timing of ET and patterns of soil moisture storage. For example, Prater and De Lucia (2006) found that early spring ET rates were higher from areas converted to cheatgrass (*Bromus tectorum*), an exotic annual grass, than for native sagebrush.

Interception by western juniper canopies can reduce the amount of precipitation reaching the ground by 20% at the edge of the canopy, 50% halfway between the canopy edge and the trunk, and 70% at the trunk (Young et al. 1984). Stem flow is low, and thus the moisture captured in western juniper canopies is lost through evaporation (Miller et al. 2005). Cutting western juniper increases soil water throughout the growing season in at least the first 2 yr posttreatment (Bates et al. 2000). We are not aware of longer-term studies evaluating the influence of western juniper management on soil moisture.

Southern Great Plains. Ashe juniper intercepts 40–50% of rainfall (Fig. 16; Hester et al. 1997; Owens et al. 2006). Transpiration from an Ashe juniper community should be greater than that from an herbaceous community because evergreen Ashe juniper canopies can transpire much of the year in the subtropical portions of their range, and plants can access water to deep depths. Mature Ashe juniper trees transpire as much as 150 $L \cdot d^{-1}$, the equivalent of about 400 $mm \cdot yr^{-1}$ (Owens and Ansley 1997). Dugas et al. (1998), using the Bowen ratio/energy balance method, compared ET between intact and cleared Ashe juniper stands. For the 2-yr period following treatment, the difference in ET was about 40 $mm \cdot yr^{-1}$, but the treatment effects on ET disappeared in the third year, by which time ET was similar in treated and untreated areas.

For honey mesquite shrublands in the southern Texas plains, water balance studies suggest that conversion of mesquite to grasslands will increase recharge 15–20 $mm \cdot yr^{-1}$ (Weltz and Blackburn 1995; Moore et al. 2008). In the Rolling Plains of Texas, honey mesquite utilizes both deep and shallow soil water (Ansley et al. 1990, 1992a, 1992b), with individual plants using 30–200 $L \cdot d^{-1}$ and plants in open

Large scale brush management programs focusing on mesquite rangelands in Texas have not resulted in increased streamflow, in spite of public perceptions to the contrary. (Photo: Bradford Wilcox)

savanna settings using more water per tree than plants in dense stands (Ansley et al. 1991, 1998). At the stand scale, ET was comparable on cleared and uncleared honey mesquite rangelands (Dugas and Mayeux 1991); hence, the potential for increasing soil recharge or water yield by reducing mesquite cover in these systems is low (Carlson et al. 1990). Honey mesquite stands in the southern Great Plains can occur on fine, montmorillonitic clay soils with high shrink–swell potential. When dry, these soils develop extensive fissures that allow rapid and deep-percolation of rainfall. Mesquite removal on these soils reduced ET and increased soil moisture by about 80 $mm \cdot yr^{-1}$ (Richardson et al. 1979).

Landscape Effects. Streamflow. Brush management is commonly presumed to increase stream flow because of assumed increases in the base flow derived from increases in groundwater recharge. This has not been widely demonstrated except at the small-watershed scale, where stream flows are generated from winter precipitation (Huxman et al. 2005). A very prominent example of enhancement of stream flow subsequent to brush management is from chaparral shrublands characterized by winter rainfall (Rowe 1948; Ingebo 1972; Davis and Pase 1977; Hibbert 1983).

Increases in stream flow of ~150% were demonstrated on a 147-ha watershed following herbicide treatment in northern Arizona. Stream flow occurred mainly as a result of winter precipitation (Baker 1984). A larger-scale watershed treatment, however, failed to generate additional stream flow (Collings and Myrick 1966). Annual water yield initially increased 20% on the herbicide-treated sagebrush sites (Sturges 1994), then returned to pretreatment levels within 11 yr as sagebrush density increased. Small-watershed studies in western Colorado indicate that runoff from summer thunderstorms was reduced following conversion of sagebrush to grass (Lusby 1979). A paired watershed study in central Oregon indicated that late season spring flow may increase as a result of juniper management (Deboodt et al. 2009).

In the Edwards Plateau of Texas, Huang et al. (2006) found that spring flow increased by about 45 $mm \cdot yr^{-1}$ following Ashe juniper removal. Studies of juniper removal on small catchments where no springs were present found surface runoff was about 20% (13 $mm \cdot yr^{-1}$) lower following root plowing, which was attributed to increased surface roughness that enhanced shallow surface storage (Richardson et al. 1979). In another study, Dugas et al. (1998) found that when juniper cover was removed by hand cutting, the treatment had little influence on surface runoff from 4- and 6-ha small catchments. Similarly, Wilcox et al. (2005) found no change in runoff following juniper removal. Paradoxically, historical stream flow records in the Edwards Plateau indicate that base flows have actually increased substantially since 1960 in spite of the fact that WPs have increased markedly since that time (Wilcox and Huang 2010). The higher base flows were attributed

FIGURE 16. Rainfall simulation is a valuable tool for understanding how woody plants alter hydrological properties on rangelands. Large-scale rainfall simulation experiments, like the one conducted here on Ashe juniper rangelands, have the advantage of being able to apply known amounts of water above the tree canopies and enable quantification of canopy interception and water and erosion dynamics at the hillslope scale. (Photo: Bradford Wilcox)

> "Projects that remove saltcedar and Russian olive with the intention of reducing ET and increasing flow in streams have produced mixed results, with most studies failing to demonstrate significant long-term changes."

FIGURE 17. Brush management is commonly applied with hopes of improving stream flow and groundwater recharge. However, studies indicating that brush management may not be achieving desired outcomes with respect to water yield are accumulating. Estimates of the economic benefits of shrub control based solely on water salvage are therefore questionable. However, it may be desirable to manage cover of nonnative shrubs, such as the tamarisk shown here, to enhance wildlife habitat, biological diversity, and soil health (Shafroth et al. 2005, 2010). (Photo: Charles Hart)

to the fact that ground cover has improved across the Edwards Plateau because of livestock destocking in the region. In the Rolling Plains of Texas, small-watershed and landscape-scale evaluations within the plains found little evidence that honey mesquite removal had an appreciable affect on stream flow (Wilcox et al. 2006, 2008a).

Early studies suggested that transpirational water loss from WPs such as saltcedar (*Tamarix* spp.) and Russian olive (*Elaeagnus angustifolia*) was substantially higher than that of native riparian vegetation. Expansion of these nonnative species along riparian corridors in the western United States was thus presumed to reduce river flows and groundwater supplies, and their removal was expected to promote stream flow and groundwater recharge (Fig. 17). However, recent studies indicate that saltcedar and Russian olive transpiration is on par with that of native species (Owens and Moore 2007), and projects that remove saltcedar and Russian olive with the intention of reducing ET and increasing flow in streams have produced mixed results, with most studies failing to demonstrate significant long-term changes (Shafroth et al. 2010).

Sediment Delivery.

There are few studies of brush management effects on sediment yield at the catchment or watershed scale. Hastings et al. (2003) found that cutting trees and spreading slash in piñon–juniper woodlands in New Mexico significantly reduced erosion from 1-ha catchments. Lusby (1979) found that shrub management reduced erosion by 80% on two 4-ha sagebrush watersheds. Such studies suggest that brush management may help curtail erosion, but additional studies and studies at larger scales are needed before broad generalizations can be made with confidence.

Wildlife Response

NRCS goals of brush management for wildlife include 1) maintaining or enhancing habitat—including threatened and endangered species, with enhancements encompassing (a) slight to substantial improvement in cover, usable space, and habitat fragmentation; (b) improvement of imbalances among and within populations; and (c) neutral effects on endangered species—and 2) improving food accessibility, quality, and quantity. The challenge in meeting these goals lies with the fact that wildlife species and functional groups vary widely in their habitat requirements (Krausman et al. this volume). In addition, and as reviewed earlier, the response of vegetation and other habitat components to brush management varies, depending on a variety of factors. Impacts—positive, neutral, or negative—of brush management on wildlife therefore depend on a variety of factors (Fig. 7). Stating that brush management maintains or enhances wildlife habitat, consequently, is an oversimplification. Goals of brush management should be stated with the interacting factors that influence impacts on specific wildlife species taken into account.

Habitat is species specific, and habitat for one species may not serve as habitat for another species or group of species (Hall et al. 1997; Krausman 2002). Clearing a large tract of sagebrush to create grassland, for example, may improve habitat for grassland birds (Reinkensmeyer et al. 2007) but destroy habitat for sagebrush obligates (Klebenow 1969; Martin 1970; Green and Flinders 1980). A fundamental concept in wildlife management is that wildlife species vary in their response

to disturbance. Northern bobwhites (*Colinus virginianus*), for example, are frequently considered "early ecological succession stage" species, whereas white-tailed deer (*Odocoileus virginianus*) are considered "midsuccession species" and grizzly bears (*Ursus arctos*) "climax" species (Bolen and Robinson 2002). This implies that bobwhites, for example, should respond positively to disturbance, whereas climax wildlife species may be negatively impacted by human-imposed disturbances, such as brush management.

Brush management may affect sexes of the same wildlife species differently (Leslie et al. 1996; Stewart et al. 2003). For example, male and female white-tailed deer selected different herbicide and fire treatments in Oklahoma (Leslie et al. 1996). Anticipated conservation benefits should be stated on the basis of the species, functional group (e.g., grassland birds, woodland birds, large mammals, small mammals, etc.) or the gender that they will benefit; broad generalizations that all wildlife will be benefited by brush management should be avoided.

Brush management may affect the same species differently, depending on seasonal use patterns of the habitat being treated. For example, thinning dense big sagebrush stands can benefit sage-grouse (*Centrocercus urophasianus*) during brood rearing (Dahlgren et al. 2006) but decrease its value as winter habitat for sage-grouse and other wildlife species (Davies et al. 2009b). Mechanical brush clearing during active nesting can destroy eggs and kill nestlings. The magnitude of the impact of brush clearing during active nesting on North American bird populations is unknown.

Wildlife species response to brush management can also vary by the species of brush. Sagebrush-obligate wildlife species are negatively impacted by reductions of sagebrush abundance (Klebenow 1969; Martin 1970; Green and Flinders 1980). However, sagebrush-obligate wildlife species benefit from control of western juniper encroaching into sagebrush communities (Miller et al. 2000; Reinkensmeyer et al. 2007).

Density and canopy cover of brush before treatment and amount of brush removed strongly influence wildlife responses to brush management. Clearing some brush in a landscape with a 100% canopy cover of WPs, for example, may benefit wildlife such as white-tailed deer (Fig. 18), whereas clearing brush in a landscape with only 25% canopy cover may be detrimental (Fulbright and Ortega-Santos 2006). In areas where the two species overlap, reducing WP canopy cover to < 50% favors mule deer (*Odocoileus hemionus*) over white-tailed deer (Wiggers and Beasom 1986; Ockenfels et al. 1991; Avey et al. 2003). Northern bobwhites use habitat patches where woody cover is ≥ 30%; therefore, reducing woody canopy cover in landscapes that marginally provide sufficient woody cover may be detrimental to bobwhites (Kopp et al. 1998; Ransom et al. 2008).

Climate and Soils Mediate Outcomes.

Variation in precipitation and soil fertility may override effects of brush management on wildlife species abundance and richness in certain cases. Nutrition, productivity, and distribution of white-tailed deer, for example, may be more strongly related to variation in precipitation than to alterations in vegetation resulting from brush management. Seventeen years after root plowing, treated sites in the eastern Rio Grande Plains of Texas were dominated by huisache (*Acacia farnesiana* [L.] Willd.; Ruthven et al. 1994). Browse species important to white-tailed deer were either absent from the huisache communities that replaced the original honey mesquite-mixed brush communities or present in greatly reduced numbers compared to the mesquite–mixed brush community. Nutritional condition and population status of white-tailed deer, however, were similar in untreated and root plowed sites. Changes in body condition, reproduction, and diet were associated with variation in precipitation rather than with plant community differences. Similarly, patch burning and grazing had little effect on white-tailed deer distribution in southern Texas because drought limited vegetation response to the treatments (Meek et al. 2008). Lack of a difference in the use of aerated and aerated and burned patches by white-tailed deer has also been attributed to lack of precipitation, which constrained forb response to the treatments (Rogers et al. 2004).

> "Anticipated impacts should consider game and non-game species; and should be tailored to specific species or functional groups."

FIGURE 18. Wildlife response to brush management is species and situation specific. Mule deer and white-tailed deer respond differently to changes in shrub cover, and the white-tailed deer depicted in this photo may respond positively when shrub cover is high but may be adversely affected by brush management imposed when shrub cover is <25%. (Photo: Tim Fulbright)

Brush management effects on wildlife food and cover vary with soil productivity (Fulbright et al. 2008). Root plowing may result in long-term loss of WPs that are important as browse for white-tailed deer on upland soils, whereas in ephemeral drainages, root-plowed sites supported brush communities similar in species composition and diversity to sites that had not been disturbed (Fulbright and Beasom 1987; Nolte et al. 1994). Ephemeral drainages receive runoff from uplands and tend to have more productive soils (Wu and Archer 2005). A possible explanation for the lack of reduction in species diversity in ephemeral drainages is that growing conditions are more favorable for the reestablishment of diverse WP species following root plowing than in upland sites.

Vegetation dynamics following brush management on fertile soils in mesic environments may follow directional change toward climax as predicted by traditional models of ecological succession. In arid or semiarid environments, however, vegetation change following disturbance may be nondirectional (Briske et al. 2005). Disturbance by brush management may push vegetation across a threshold to a different plant community than existed before treatment and one that is relatively stable. This new plant community may or may not provide better-

quality habitat for specific wildlife species than the plant community that existed before brush management. For example, exotic annual grasses can rapidly increase and dominate plant communities after brush management in the intermountain West (Stewart and Hull 1949; Evans and Young 1985; Young and Allen 1997). Nonnative annual grass invasion in sagebrush communities decreases their habitat value for sagebrush-obligate and facultative wildlife (Davies and Svejcar 2008). Buffelgrass may increase following root plowing or disking in South Texas (Gonzalez and Dodd 1979; Johnson and Fulbright 2008) with adverse effects on bobwhite populations (Flanders et al. 2006). Thus, the potential for undesirable shifts in plant communities following brush management must be carefully considered before implementing treatments (see the sections "Biodiversity and Nonnative Species" and "A Tool to Promote Landscape Heterogeneity and Biodiversity").

Scale and Pattern. Effects of brush management on wildlife may vary dramatically, depending on scale of application (see also Bestelmeyer et al. this volume). Many grassland-adapted species may respond in a positive fashion to broad-scale conversion of woodland to grassland (Fitzgerald and Tanner 1992; Smythe and Haukos 2010). Conversely, these large-scale conversions reduce northern bobwhite and Texas tortoise populations (Kazmaier et al. 2001; Ransom et al. 2008). Extensive brush removal (>60% of the landscape) reduces landscape use by white-tailed deer (Rollins et al. 1988; Reynolds et al. 1992). Large areas of untreated brush provide habitat for many nongame bird species, and brush management efforts should be limited in scope in areas where conservation of this wildlife group is a priority (Fulbright and Guthery 1996).

Range management has traditionally promoted vegetation uniformity rather than heterogeneity (Fuhlendorf and Engle 2004). Promoting uniformity, deemed prudent for increasing livestock production, included practices such as clearing WPs completely from the landscape, planting monotypic stands of grasses, and taking steps to promote livestock grazing distribution. Wildlife needs were relegated to lesser importance in this traditional management approach. Wildlife response to amount and interspersion of brush patches varies among species. Many wildlife species reach maximum diversity or density in heterogeneous landscapes such as those containing a mosaic of brush and interspersed tracts dominated by herbaceous vegetation (Roth 1976; Tews et al. 2004a; see also the sections "Biodiversity," "Biodiversity Response," and "Biodiversity and Nonnative Species"). Diversity and richness of birds is greatest in plant communities with structural heterogeneity (Reinkensmeyer et al. 2007). For example, providing a mosaic of plant communities including closed-canopy oak forest and open pastures derived from forest increased breeding nongame birds richness in Oklahoma (Schulz et al. 1992). Brush management is commonly done in strips or other patterns to create mosaics of WP communities interspersed with communities dominated by herbaceous plants to benefit wildlife (Fulbright and Ortega-Santos 2006). Brush sculpting is another approach to brush management (Fulbright 1997; McGinty and Ueckert 2001). Brush sculpting refers to selective removal of brush to accomplish multiple-use objectives, such as habitat improvement for wildlife and increased forage for livestock (Ansley et al. 2003). Anticipated effects of brush management should take into account the extent to which habitat heterogeneity is important for wildlife species (Fulbright 1996; Kie et al. 2002; Tews et al. 2004a).

Patch size is also an important consideration when creating vegetation mosaics (Bestelmeyer et al. this volume). Selection of patch size depends on management objectives and the wildlife species or functional group being managed. Mosaics may be created to either maximize wildlife species diversity or optimize habitat for a particular species. Edge and interior species are more prone to be affected by patch size than are generalist species (Bender et al. 1998). A mosaic consisting of patches that are too small essentially functions as edge and does not provide habitat for interior species. For woodland-adapted birds, patch size and shape are important because nest parasitism and nest predation may increase with increasing edge, although this relationship has been questioned in recent literature (Patton

"Vegetation mosaics may be created to either maximize wildlife species diversity or optimize habitat for a particular species."

1994; Lahti 2001). Patches that are large with relatively little perimeter support fewer edge species.

Patch size and configuration requirements vary among wildlife species. Grassland birds, for example, require patches >50 ha (Helzer and Jelinski 1999). Ratio of patch perimeter to area is also important; bird species richness is greatest in patches with larger interiors that are free from edge effects. For grassland birds, landscape composition may interact with patch size in that larger core areas may be more important in landscapes with a mixture of grassland and woodland than in treeless landscapes (Winter et al. 2006).

Although the idea of creating patchy mosaics through brush management has been discussed in the literature, using brush management to achieve an "optimum" size and configuration of patches has received little attention (Fulbright 1996). Part of the reason for the lack of attention to the concept of optimal patch size/configuration may be that many of the game species that are often the focus of research on brush management effects are edge associates that show little response to variation in patch size. Northern bobwhites, for example, appear to be adapted to an almost infinite set of patch configurations; therefore, an "optimum" arrangement may not exist (Guthery 1999).

Brush management may increase connectivity and reduce habitat fragmentation for grassland-adapted species; conversely, brush management may fragment habitat of shrubland or woodland adapted species if the cleared areas limit wildlife movement between tracts of woody vegetation. Patches of habitat for a wildlife species should be linked by corridors that facilitate movements among habitat patches (Bennett 2003). Ensuring that connectivity exists among habitat patches should be a priority when vegetation is manipulated.

Improving Food. Brush management may improve food accessibility, quality, and quantity for some wildlife species or functional groups (e.g., grazers) but reduce it for others (e.g., browsers). A review of publications in the *Journal of Range Management*, *Rangeland Ecology & Management*, and *Ecology* and articles emerging from a search of BIOONE, JSTOR, Science Direct, and Springer using the search strings "brush management," "brush management wildlife," "herbicides birds," "brush control deer," "brush control prairie chicken," and "brush control sage grouse" yielded 50 articles addressing 59 cases of effects of brush management treatments or combinations of treatments (e.g., fire and herbicides) on wildlife food plants. Effects on food plants ranged from positive (53%) to neutral (32%) to negative (16%). In most cases, negative responses occurred where brush management reduced mistletoe (a parasitic plant on honey mesquite that is eaten by

Mechanically clearing juniper in strips provides edge and brush piles for wildlife, forage for livestock and opportunities for future use of prescribed fire as a management tool. (Photo: Kirk McDaniel)

Plant communities with mixtures of herbaceous- and shrub-dominated patches provide excellent habitat for a diversity of game and non-game species. (Photo: Tim Fulbright)

deer), reduced browse plants preferred by white-tailed deer, or increased thorns or secondary compounds in browse regrowth. In the review, we considered treatment effects to be neutral when they resulted in only temporary (<3 yr) increases in forb seeds or insects. Chemical, mechanical, and pyric brush management methods vary in their impact on woody and herbaceous food for wildlife. Chemical treatments, for example, tend to cause a temporary reduction in forbs, whereas fire may stimulate growth and abundance of early successional forbs that benefit many species of animals (e.g., Fig. 13) (Beasom and Scifres 1977; Bozzo et al. 1992a). Fire may top kill WPs, encouraging production of palatable sprouts (Schindler et al. 2004b). Anticipated benefits of brush management to wildlife should be predicated on the brush management approach to be used and the wildlife species potentially affected.

Endangered Species. Brush management potentially reduces habitat for endangered species that depend on WP communities, such as ocelots (*Felis pardalis*), which need woodland with >97% canopy cover, or pygmy rabbits (*Brachylagus idahoensis*), which forage primarily on big sagebrush (Green and Flinders 1980; Harveson et al. 2004). Conversely, brush management potentially could improve habitat for grassland-adapted species, such as the Attwater's greater prairie chicken (*Tympanuchus cupido attwateri*). Documentation of the effects of brush management on habitat of species listed as endangered in the United States is lacking, however.

Herbicide Toxicity. Herbicides used in rangeland brush management are usually not used in concentrations harmful to wildlife and dissipate from the ecosystem following the growing season they are applied (Scifres 1977; Freemark and Boutin 1995; Guynn et

Wyoming big sagebrush mowed in strips creates a mosaic of treated and untreated sagebrush habitat to increase diversity and maintain critical habitat for sagebrush-obligate wildlife. (Photo: K. W. Davies)

al. 2004). Herbicides are generally not acutely toxic to soil organisms (Freemark and Boutin 1995). Certain aspects of herbicide toxicity to wildlife, such as the role of surfactants and inert ingredients, and possible synergistic effects of multiple chemicals applied simultaneously are unknown (Guynn et al. 2004). Herbicides may negatively affect insects directly or indirectly, but little is known of the effects of rangeland herbicides on these organisms. A better understanding is needed since native rangelands may serve as a reservoir of pollinator and predator insects important to crop production in nearby cultivated areas (Freemark and Boutin 1995). In addition, invertebrates are a critically important food resource for many grassland bird species (O'Leske et al. 1997). Research on herbicide effects on reptiles and amphibians is also lacking (Freemark and Boutin 1995; Guynn et al. 2004).

Although rangeland herbicides are generally not highly toxic to wildlife, acute effects of the herbicide 2,4-D have been documented. The herbicide is toxic to cutthroat trout (*Salmon clarkia*) (Woodward 1982). Spraying 2,4-D dramatically reduced pocket gopher (*Thamomys talpoidis*) populations in Colorado (Keith et al. 1959).

Predators. Anticipated benefits of brush management stated by NRCS focus largely on forage production and habitat structure for herbivores; however, brush management also alters predator habitat and may change behavioral responses of prey. Ungulates, for example, may use cleared patches within woodland or shrubland because of enhanced visual detection of predators (Bozzo et al. 1992b). Florida panthers (*Felis concolor coryi*) are attracted to recent prescribed burns where prey species such as white-tailed deer congregate (Dees et al. 2001). Landscape-level reduction of brush may remove perching structures important for raptors and increase susceptibility to nest predators. Prickly pear (*Opuntia* spp.) control, for example, has the potential to reduce nest sites and increase nest susceptibility to predators for bird species that prefer nesting

in prickly pear. Treating prickly pear with herbicides, however, did not reduce nesting success of bobwhites in central Texas (Hernandez et al. 2003). Prey population densities may also change in response to brush management. Effects of mechanical brush management on the mortality of small mammals and immobile wildlife species at the time of treatment are unknown. Habitat changes following treatment may have unintended consequences, such as favoring increased prey densities. For example, cotton rat (*Sigmodon hispidus*) densities were six times greater on root-plowed rangeland in Texas than in untreated rangeland (Guthery et al. 1979). Rodent populations are strongly cyclical. Flushes in rodent abundance may be followed by increases in predator abundance; but subsequent abrupt declines in rodent populations may cause the now-abundant predators to shift to a prey base of livestock or ungulates such as white-tailed deer.

Brush management may also affect visual cues used by predators to locate prey. Logged areas in the boreal forests of Canada have less debris on the forest floor than uncut stands. Efficiency of predation by martens (*Martes americana*) is greater in uncut timber stands because coarse woody debris act as sensory cues and enhance hunting success (Andruskiw et al. 2008). Brush management may likewise affect structure and amounts of woody debris in shrubland habitats, potentially affecting predator efficiency. Herbicide application may have little influence on habitat use by coyotes (*Canis latrans*) and bobcats (*Felis rufus*) possibly because standing woody material remains after treatment and herbaceous community structure is not drastically altered (Bradley and Fagre 1988).

Treatment Longevity. Brush management initially reduces shrub canopy cover, but over time, stem and foliage cover returns. In Texas, the estimated duration of treatments range from 10 yr to 20 yr for root plowing and from 3 yr to 9 yr for roller chopping (Fulbright and Taylor 2001; Schindler and Fulbright 2003). Potential benefits of brush management for wildlife, therefore, are transient. Brush management, for example, may benefit a wildlife species initially, but as the WP community reestablishes (e.g., Fig. 10), benefits may be lost. The temporary nature of treatments and the need for follow-up treatments must therefore be explicitly considered in statements of anticipated benefits (see the previous sections "Integrated Brush Management Systems" and "Treatment Options").

Single applications of mechanical brush management with no follow-up treatments may adversely impact wildlife habitat. For example, density of WPs palatable to white-tailed deer may be lower in WP communities that reestablish following root plowing than in undisturbed communities (Fulbright and Beasom 1987). Density of woody legumes such as honey mesquite and huisache may be greater on root plowed areas than on untreated areas >17 yr posttreatment (Fulbright and Beasom 1987; Ruthven et al. 1994). WPs that regenerate following roller chopping may have longer and more numerous spines than undisturbed plants, which could reduce bite rate of browsers (Schindler and Fulbright 2003; Schindler et al. 2004a).

Measuring Habitat Improvement. The statement of anticipated benefits of brush management to wildlife is based on the assumption that improvements in food, cover, space, imbalance among populations, and fragmentation are evidence of habitat improvement. Vegetation characteristics are commonly linked with habitat quality in the wildlife literature (Guthery 1997; Hall et al. 1997; but see Johnson 2007). However, increases in a specific habitat characteristic do not constitute improvement if that characteristic is not limiting to wildlife (Guthery 1997). For example, rangeland disking may increase abundance of seed-producing forbs. However, seeds may not be limiting to northern bobwhites (Guthery 1997). In this case, the assumption that increasing food (e.g., seed-producing forbs) resulted in habitat improvement may not be valid. Further, numerous confounding factors exist in natural ecosystems, and an increase in food and cover alone may not result in habitat improvement if some other factor, such as nesting cover, is limiting.

Brush management is assumed to have improved wildlife habitat quality in an area

if it results in greater food abundance, better interspersion of plant communities, and habitat requirements, less fragmentation, or better cover characteristics. An underlying assumption is that population density in an area increases with increasing habitat quality (Guthery 1997). However, increased densities following brush management does not necessarily indicate sustained improvement in habitat. Treated areas may provide resources needed by an organism only during part of the year, and untreated areas may be needed to meet needs during other times of the year. White-tailed deer, for example, do not exhibit preference for a particular level of woody canopy cover during winter, but during summer, deer densities increase with increasing WP cover, with areas >80% canopy cover receiving greatest use (Steuter and Wright 1980). Improvements in habitat quality should be expressed in terms of increased survival and reproduction in addition to increased population densities and availability of key habitat components (Van Horne 1983; Hall et al. 1997; Crawford et al. 2004). For northern bobwhites, evidence that their abundance increases with habitat quality variables such as food supplies and interspersion is limited and equivocal (Guthery 1997). Instead, abundance of bobwhites is proportional to the amount of usable space (habitat for which a species is fully adapted), and only practices that increase the abundance of usable space are likely to improve bobwhite numbers (Guthery 1997; Guthery et al. 2005). The usable space concept has also been applied to white-tailed deer management (Hiller et al. 2009).

Demographic characteristics of wildlife populations and usable space are more difficult and time consuming to quantify than habitat characteristics such as food production. As a result, comparisons of survival

Brush management may improve food accessibility, quality, and quantity for some wildlife species or functional groups but reduce it for others. (Photo: Tim Fulbright)

and reproduction of wildlife on sites with and without brush management are limited (Appendix II). Consequently, restricting statements of anticipated benefits to treatments and species for which increased reproduction, survival, and density or increases in usable space resulting from brush management have been documented is impractical. A better approach would be to acknowledge that while brush management may improve various habitat properties, its impact on habitat quality for many species is unclear.

Fuels Management

Brush management is increasingly being applied in shrubland and woodland settings to reduce fire risk or create fuel breaks (Keeley 2002; Davies et al. 2009b); however, little information is available to evaluate its effectiveness. In forest systems, mechanical brush management alters fuel characteristics and influences fire behavior (Kane et al. 2009); however, current fire models have not yet been parameterized to represent these modified behaviors. Although the impact of brush management on fire characteristics and spread are unclear, fire suppression efforts can be facilitated simply by reducing fuel height (Keeley 2002). However, while brush management can effectively reduce the mass and continuity of canopy fuels, it may promote production and continuity of fine surface fuels (e.g., grasses) and thus promote fire risk (Keeley 2002; Perchemlides et al. 2008; Huffman et al. 2009).

RECOMMENDATIONS

- Care is needed when using words and phrases such as "vigor," "health," "biodiversity," "encouraging growth," and "suitable" when projecting the effects of brush management. These terms are vague or ill-defined and often value laden and should be replaced with words and phrases that refer to specific and tractable metrics to define more specific and measurable conservation outcomes.
- Integrated Brush Management Systems have proven effective in WP management and are likely to yield the greatest conservation benefits. Brush management is a long-term commitment. Adaptive management, coordination with grazing management, a plan and funding for follow-up restoration and brush treatments, and periodic monitoring are essential. Emphasize flexibility and objectivity.
- Customize brush management prescriptions according to the stakeholder's vision and management objectives and the inherent capability or limitations of the ecological site. This perspective on human dimensions should be incorporated into the list of purposes in practice code 314: "Work closely and cooperatively with clientele to apply brush management practices that meet both land and personal conservation objectives."
- Evaluate and define when, where, how, and under what circumstances brush management should be undertaken and what specific outcomes are to be attained. Recommendations should be thoroughly vetted and justified. Do not assume that brush management is needed simply because shrubs are present.
- Tailor statements of potential hydrological benefits of brush management to specific bioclimatic zones.
- Anticipated effects of brush management should take into account the extent to which habitat heterogeneity is important for wildlife species. Do not assume that brush management will result in improvement of habitat for a wildlife species or functional group. Tailor statements of anticipated benefits of brush management to specific habitat variables or characteristics, such as food production, and to specific wildlife species or functional groups. State which wildlife species or functional groups may be negatively impacted by brush management under specific sets of circumstances.
- Develop and maintain a relational database to evaluate brush management treatments. Important information may include treatment approaches and longevities; location and spatial pattern(s) of treatment in relation to soils and topography; pre- and posttreatment soil, plant, livestock, and wildlife responses; environmental conditions; and predicted trade-offs and outcomes based on published literature (Table 6). This database should be updated as new information becomes available and used to communicate anticipated benefits for specific locations and regions.

TABLE 6. Example of a matrix approach to communicating anticipated benefits of brush management for wildlife. A similar matrix could be developed for plants, soils, and so on.

Brush management approach	Scale	Climate	Existing woody canopy cover (%)	Wildlife species or group	Anticipated impact[1]
Mechanical	Landscape	Humid	60–100	Grassland obligates	+
				Woodland obligates	–
				Edge-associated species	–
				Habitat generalists	0
			25–59	Grassland obligates	+
				Woodland obligates	–
				Edge-associated species	–
				Habitat generalists	0
			<25	Grassland obligates	+
				Woodland obligates	–
				Edge-associated species	–
				Habitat generalists	0
Chemical, fire	Mosaic, patch	Subhumid, semiarid, arid	<25	Habitat generalists	

[1]+, positive; 0, neutral; –, negative.

- Seeding of nonnative plants following brush management should be avoided, but if considered, it should be explicitly justified.
- Articulate and critically evaluate positive and negative trade-offs in brush management impacts on various ecosystem goods and services. For example, gains in livestock production and herbaceous diversity accruing from brush management may be at the expense of ecosystem carbon sequestration.
- Develop a mechanism to integrate conservation planning on individual properties into and consistent with local/regional conservation plans. Specific goals and objectives from brush management may vary by ownership and agency, but by pooling expertise and financial resources, there will be better opportunities for treating and restoring larger areas.

KNOWLEDGE GAPS

- The extent to which pre–brush treatment management conditions drive posttreatment responses is largely unknown, as are the effects of follow-up treatments.
- Projected effects of brush management mention numerous variables related to air quality as "not applicable." However, available information, albeit scant, suggests that changes from grass to WP dominance can significantly increase emissions of trace gases and volatile organic carbon compounds and the production of dust, aerosols and allergens. The extent to which brush management might reverse these is unknown, as are the implications for human health, tropospheric chemistry, and land surface–atmosphere interactions.
- ANPP can be dramatically enhanced by shrub encroachment (Knapp et al. 2008a; Barger et al. 2011), but the effects of brush management on ANPP are largely unknown. Plant production responses to brush management have focused on the herbaceous vegetation, and there is scant data on WP ANPP during the postmanagement period. Thus, we are ill equipped to evaluate brush management from a carbon-accounting perspective.
- The belowground organic carbon pool (roots+soil) typically dwarfs the aboveground pool in rangeland ecosystems. Robust generalizations as to how WP encroachment (Fig. 4) and brush management affect this large belowground

pool are not yet possible. Studies that have quantified soil responses to brush management are few (Fig. 8) and have relied on comparing random samples from a treated site(s) to a nearby, untreated site. Given the extensive edaphic heterogeneity on shrub-encroached rangelands (e.g., Bai et al. 2008; Liu et al. 2010), such coarse comparisons are probably not too reliable. Studies quantifying soil resources in a spatially explicit fashion before and following brush management are sorely needed, as are studies quantifying the response of shrub roots to brush management. Decreases in plant and SOC pools that may occur following brush management could have important but as yet poorly understood implications for ecosystem carbon management.

- Quantification of trade-offs between livestock production, hydrology, erosion, carbon sequestration, biodiversity, and so on and approaches for weighting them is a current challenge that must be addressed if we are to advance our ability to comprehensively evaluate the conservation value of brush management. Brush management has the potential to modify the provisioning of numerous ecosystem services at both local and regional scales. Attempts must be made to monitor and value these nontraditional nonmarket services.
- Many of the potential benefits of brush management depend on the extent to which herbaceous production and ground cover can be reestablished and the duration of the herbaceous response. General models of WP effects on herbaceous vegetation (Fig. 3) need to be better quantified to determine when it might be most effective to implement brush management, and conceptual models of posttreatment herbaceous vegetation response to brush management (Fig. 10) need to be made operational to obtain quantitative ecological (Fig. 9) and socioeconomic (Fig. 11) assessments of

Woody plant encroachment represents a threat to grassland, shrub-steppe, and savanna ecosystems and the plants and animals endemic to them. (Photo: Tim Fulbright)

brush management. Simulation modeling has been underutilized (Fig. 8). Given the advent of inexpensive, user-friendly software for personal computers, this tool can now be readily used to integrate existing information for assessment, scenario development, and forecasting (e.g., Grant et al. 1999; Fuhlendorf et al. 2008).

- The major knowledge gap related to brush management and water is our limited understanding of landscape-level implications. With the exception of a few studies (e.g., Collings and Myrick 1966; Wilcox et al. 2008a), there has been little documentation of the large-scale impacts of brush management on water and erosion processes. As a result, there is considerable uncertainty concerning the efficacy of extrapolating from fine-scale studies to the landscape level (Wilcox and Huang 2010).
- Biodiversity responses to shrub encroachment are poorly documented, and responses to brush management have focused largely on herbaceous vegetation. Responses of various faunal groups, including soil biota, are few and scattered. The implications of changes in biodiversity for ecosystem function have been the topic of much discussion in the research community but remain poorly understood.
- Brush management effects on wildlife have focused mainly on game species, particularly white-tailed deer, northern bobwhites, and sage-grouse. Nongame species, including predators, passerines, small mammals, and reptiles, have been largely neglected. Habitat requirements of many nongame species are not well understood, making it challenging to even speculate about effects of brush management. These gaps must be filled for statements of anticipated benefits to be made for specific species or functional groups.
- The extent to which brush management-induced changes in habitat attributes translate into improvements in carrying capacity and animal birth rates, longevity, nutritional status and body mass are largely unknown.
- Further research that addresses the interrelationship between brush management and fire behavior is needed to provide robust conclusions on its effectiveness for reducing fire risk and spread. Trade-offs between reducing WP canopy mass and continuity and promoting fine fuel production needs further study among different WP communities.
- A framework for conceptualizing how climate change, invasions of nonnative species, and increases in atmospheric CO_2 and nitrogen deposition might influence future grass–woody states and ecosystem responses to brush management is needed.

CONCLUSIONS

Successful long-term management programs (typically >5 yr) usually involve an integrated

Rangeland conservation goes beyond traditional concerns of livestock production to include potential effects on a variety of ecosystem services. The research community is challenged with measuring and monitoring these varied impacts; and the management community with creating or maintaining woody-herbaceous mixtures in arrangements that satisfy competing objectives. (Photo: Tim Fulbright)

brush management systems and restoration approach that includes a suite of mechanical, fire, biological, and chemical methods. A combination of methods customized for local ecological site conditions is particularly important when the primary objective is to achieve long-term native plant stability that supports conservation and resource function.

Assessing revegetation potential is a critical first step before proceeding with brush management. Brush management and revegetation costs are high, and careful selection of areas with a high potential for reestablishment is necessary for long-term, sustainable brush management. In many situations, herbaceous vegetation on treated areas will recover naturally after brush management without revegetation. In other situations, planting or seeding of grasses or forbs may be necessary. Sites with particularly dense brush cover, poor hydrologic integrity, or related conditions may have limited revegetation potential. An in-field evaluation and soil survey should always be used to evaluate soil and other factors that will ultimately influence replacement of the vegetation community. With these caveats in mind, our synthesis suggests the following conclusions regarding the conservation value of brush management:

- Conservation of grasslands and savannas as ecosystem types and the plants and animals endemic to them should be a high priority (Fig. 19). Loss of grassland-obligate organisms occurs with shrub encroachment, even if overall numerical biological diversity is enhanced or unaffected. Brush management programs are essential to maintain grassland, steppe, and savanna ecosystems and the biodiversity and services they provide. Progressive brush management protocols will be required to achieve this conservation goal in many instances.
- Herbaceous cover, production, and diversity are typically enhanced by brush management. However, exceptions occur, and the possibility for deleterious outcomes should always be anticipated and considered when planning. Furthermore, treatment longevity will vary, so plans for follow-up are required.

FIGURE 19. Brush encroachment threatens habitat for grassland-obligate species such as this savanna sparrow. Brush management may be required to generate and maintain shrub cover amounts and patterns within acceptable limits for such species. (Photo: Tim Fulbright)

- Returns arising from improved livestock performance and production are important, but benefits beyond livestock production are being increasingly recognized. When the value of ecosystem goods and services beyond those associated with livestock production are taken into account, a more favorable picture of brush management begins to emerge.
- Although frequently justified on the basis of benefits to water quality and quantity, brush management does not necessarily produce the hydrological benefits that are commonly attributed to it. In most cases, these perceived benefits are exaggerated and have not been documented, and there is little or no evidence that brush management is a viable strategy for increasing ground water recharge or stream flows at meaningful scales. Outcomes depend on the vegetation type and geological setting. In some cases, depending on the vegetation community and the method of shrub management, surface runoff and erosion may actually increase. Local/regional knowledge should therefore guide brush management prescriptions with respect to hydrological impacts. In settings where winter precipitation predominates or where WPs are accessing deep stores of water, there is the potential to use vegetation management to enhance groundwater recharge and stream flow. However,

A burned (left) and untreated (right) mountain big sagebrush plant community on the Hart Mountain National Wildlife Refuge in southeastern Oregon. (Photo: K. W. Davies)

projections for how this translates to watershed- and regional-scale hydrology is based more on speculation than data.

- Statements that brush management maintains or enhances wildlife habitat are oversimplifications. Habitat requirements of many nongame species are poorly understood, making it challenging to even speculate about effects of brush management on these organisms. Clearer definitions of what constitutes a benefit of brush management to wildlife are needed, and these should be tailored to species or functional groups. Statements should focus on the habitat characteristics or attributes that are anticipated to be improved.

Technology and the tools available for brush management are dynamic and ever changing. Keeping educated and up to date on new developments is paramount. There are knowledge gaps in brush management, but there always will be, and it is important that managers strive to use the best available information. In some instances, practices applied and approaches followed to manage a particular WP species may

not be known. Thus, it is recognized that land managers are often placed in situations where they must exercise flexibility, responsibility, and their best professional judgment when developing a planning strategy and carrying out an action program.

Brush management presents a series of dilemmas and challenges as a response to WP encroachment. The recognition that WP proliferation can substantially promote ecosystem primary production and carbon stocks may trigger new land use drivers as industries seek opportunities to acquire and accumulate carbon credits to offset CO_2 emissions. WP proliferation in grasslands and savannas may therefore shift from being an economic liability in the context of livestock production to a source of income in a carbon sequestration context. Policy and management issues related to grazing land conservation thus extend well beyond the traditional concerns of livestock production and game management (wildlife valued for sport hunting) to include potential effects on hydrology, carbon sequestration, biological diversity, atmospheric chemistry, and the climate system. The research community is challenged with quantifying and monitoring these varied impacts and the management community with devising approaches for creating or maintaining woody–herbaceous mixtures in arrangements that satisfy competing conservation objectives.

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APPENDIX I. Herbaceous Response to Brush Management

A search of articles with the key word "brush control" in the *Journal of Range Management* and *Rangeland Ecology & Management* at http://www.uair.arizona.edu/search?page_set = 51 yielded 1277 separate articles. Among these, about 80% (1021 articles) either assessed target plant mortality (764 articles) or described changes in herbaceous (grass and broadleaf) plant abundance (e.g., cover, biomass, and frequency; 257 articles). Of the 257 articles reporting on herbaceous responses, 216 (84%) characterized the response as positive, 21 (8%) reported no change, and 20 (8%) report a negative response to brush management.

In another, more directed search, we sampled published accounts of how brush management influences herbaceous vegetation. Web of Knowledge searches resulted in 532 unique references, 36 of which were field studies conducted on rangelands in the United States and 22 of which measured the response of herbaceous or grass production.

Among these 22 studies, herbicide was the most frequently assessed brush management technique (15 studies, or 68%). As with our initial, broader survey, most of these (18 studies, or 82%) reported increases in herbaceous production. The majority of experiments were conducted over short periods of time, with only eight studies (36%) lasting more than 5 yr and only five (23%) lasting longer than 10 yr.

APPENDIX II. Brush Management and Wildlife Habitat Quality

- Peer-reviewed publications were surveyed to determine the proportion of studies that measured effects of brush management on wildlife density.
- A total of 97 publications emerged in this compilation, which included articles in the *Journal of Range Management*, *Rangeland Ecology & Management*, and *Ecology*, along with those emerging using the search strings "brush management," "brush management wildlife," "herbicides birds," "brush control deer," "brush control prairie chicken," "brush control sage grouse," "fire sagebrush," "sage grouse prescribed fire," and "prescribed fire" in the search engines BIOONE, JSTOR, Science Direct, and Springer.
- Only 45% of these articles reported some measure of organism abundance in response to brush management.
- Only about 5% reported the demographic information that Van Horne (1983) and Hall et al. (1997) suggest as necessary to assess habitat quality.

APPENDIX III. Citations for Data Points in Figures 4, 9, and 13

Data points in Figure 4 are from the following: 1–3 = Schlesinger and Pilmanis (1998); 4–5 = Asner et al. (2003); 6, 8–12, 17 = Geesing et al. (2000); 7 = Hughes et al. (2006); 13–15 = Boutton et al. (1998); 16 = Tilman et al. (2000); 18–19 = Mordelet et al. (1993); 20 = San Jose et al. (1998); 21–24 = Wheeler et al. (2007); and 25–34 = Jackson et al. (2002).

Data points in Figure 9 are from Ansley et al. (2006), Bedunah and Sosebee (1984), Clary (1971), Griffith et al. (1985), McDaniel et al. (1982), Morton et al. (1990; mechanical treatments), Augustine and Milchunas (2009), Bates et al. (2005, 2009), Cable (1967), Engle et al. 1993, 1998), Teague et al. (2008b; prescribed fire), Bedunah and Sosebee (1984), McDaniel et al. (1982), Morton et al. (1990; herbicides), and Engle et al. (1993; multiple treatments).

Data points in Figure 13 are from Baeza and Vallejo (2008), Davies et al. (2007), Edwards et al. (2007), Maccherini et al. (2007), Maron and Jefferies (2001), Nolte and Fulbright (1997), Nolte et al. (1994), Olson and Whitson (2002), Page et al. (2000), Ponzio et al. (2006), Ruthven et al. (1993), Ruthven and Krakauer (2004), and Sheley et al. (2006).

CHAPTER

4

Assessment of Range Planting as a Conservation Practice

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“The goal of a conservation management plan is to transition an existing undesirable plant community to a more desirable state.”

Assessment of Range Planting as a Conservation Practice

4

Stuart P. Hardegree, Thomas A. Jones, Bruce A. Roundy, Nancy L. Shaw, and Thomas A. Monaco

INTRODUCTION

The Range Planting Conservation Practice Standard is used to inform development of Natural Resource Conservation Service (NRCS) management recommendations for improving vegetation composition and productivity of grazed plant communities. Range planting recommendations are generally implemented within an integrated conservation management system in conjunction with related conservation practices such as brush management, prescribed burning, prescribed grazing, herbaceous weed control, and upland wildlife habitat management. The Range Planting Standard is defined as "establishment of adapted perennial or self-sustaining vegetation such as grasses, forbs, legumes, shrubs and trees." The six specific purposes of this standard are to:

- Restore a plant community similar to the Ecological Site Description reference state for the site or the desired plant community.
- Provide or improve forages for livestock.
- Provide or improve forage, browse, or cover for wildlife.
- Reduce erosion by wind and/or water.
- Improve water quality and quantity.
- Increase carbon sequestration

Additional conservation effects associated with related conservation practices include reduction of negative weed impacts and reduction of wildfire hazard. Range planting conservation practices apply where desirable vegetation is below the acceptable level for natural reseeding to occur, or where the potential for vegetation enhancement by grazing management is not satisfactory.

Range planting was implemented on 517 000 ha of grazing land in the 17 conterminous western states in the 5-yr period, 2004–2008. This is a relatively small area compared to implementation of some other conservation practices within the region over the same period: prescribed grazing, 31 360 000 ha; upland wildlife habitat management, 19 166 000 ha; herbaceous weed control, 7 603 000 ha; brush management, 1 457 000 ha; and prescribed burning, 371 000 ha. Conservation cover, a closely related conservation practice for reducing erosion on retired cropland as part of the Conservation Reserve Program (Young and Osborn 1990), was implemented on over 1 600 000 ha during the same period.

Site-specific conservation management plans are developed for areas where existing plant community attributes are insufficient to meet management goals for productivity or species composition, and where natural recovery toward a more desirable state is not expected. The goal of a conservation management plan is to transition an existing undesirable plant community to a more desirable state. It is assumed that successful implementation of this change in state will be associated with specific conservation effects.

The National Standard for range planting is usually modified at the state level with specific recommendations of regional or local relevance. State standards, however, retain the same general guidance, and usually vary only in the degree to which they include more detailed recommendations extracted from region-specific NRCS technical notes and seeding guides. Recommendations for the following management elements are common to both the national and state range planting standards: selection of appropriate plant materials,

Rangeland drill used for site preparation and seed placement. (Photo: USFS, 2006)

USDA-ARS Plant material selection trials, Beaver, UT (Photo: Craig Rigby, 2009)

seed-bed preparation, planting methods, seeding depth, seeding rate, time of seeding, postplanting management, and weed control.

The spatial domain of interest in this evidence-based assessment includes rangeland systems in the Great Plains, Intermountain West, southwestern desert, and interior-California hydroclimatic zones (Barbour and Billings 2000). These areas are characterized by different vegetation types, management priorities, and climatic syndromes that also vary internally along both latitudinal and elevational gradients (Natural Resources Conservation Service 2006). Range planting issues common to all areas are a generally arid or semiarid climatology, and high annual and seasonal variability in weather and climate. These areas are also commonly under pressure from highly competitive annual and perennial weeds or expanding populations of native woody plants.

The success of specific conservation practice recommendations and the potential ecological outcomes realized are both highly dependent upon ecological site characteristics, the initial degree of deviation from desired site characteristics, and weather, all of which are highly variable in both time and space. An important implication of this high variability, in both initial establishment and later-seral processes, is that virtually no direct experimental evidence exists to link specific range planting conservation practices to conservation effects *per se*. The linkage is instead derived indirectly through evidence of

the degree to which specific planting techniques have been shown to produce successful plant establishment, and evidence supporting the positive conservation effects of alternative vegetation states. We have, therefore, separated our assessment of the conservation effects of rangeland planting practices into two components: assessment of the direct benefits of specific planting techniques recommended in the range planting standard, and assessment of specific conservation effects of alternative vegetation states. The assessment of rangeland planting techniques involved a survey of 189 range planting studies from the refereed journal literature. These studies were classified as to bioclimatic zone, initial plant community and type of disturbance, plant materials and seed-mix characteristics, seeding rate, site preparation and weed control methodology, planting depth, planting season, experimental design, weather, and relative success criteria. Summary statistics cited in this synthesis were derived from the survey.

ASSESSMENT OF THE DIRECT BENEFITS OF RANGE PLANTING PRACTICES

The range planting standard specifically requires selection of plant materials that are adapted to both climate and microclimate as affected by soil type, landscape position, and range site characteristics. Gross climatic variability generally determines the historical complement of native species at a site, but also the suitability of introduced plant materials (Shown et al. 1969; Shiflet 1994; Vogel et al. 2005; Natural Resources Conservation Service 2006). Seedbed preparation and planting methods are designed to optimize microclimatic conditions for planted species, to increase the number of favorable microsites for germination and establishment, and to mitigate or control competition from undesirable species (Roundy and Call 1988; Call and Roundy 1991; Sheley et al. 1996; Krueger-Mangold et al. 2006; Sheley et al. 2006).

A major problem with synthesizing range planting research results is the high variability in metrics used to evaluate success. Relatively few authors have directly evaluated alternative criteria for quantification of success (Ries and Svejcar 1991). The majority of range planting studies use arbitrary, relative criteria for judging success, and only consider planting-year or first-year effects. Typical criteria for evaluating success generally involve measurements or ocular estimates of density, frequency, cover, and/or biomass.

SELECTION OF PLANT MATERIALS

Climatic Considerations

Weather and climate patterns in western North America are highly variable in space and time (Rajagopalan and Lall 1998). The relationship between climate and both vegetation distribution and production on western rangelands is well documented (Barbour and Billings 2000; Natural Resources Conservation Service 2006). The general importance of climate is acknowledged in seeding guides in the form of tables that list species and cultivar suitability as a function of mean annual precipitation (Jordan 1981; Jensen et al. 2001; Lambert 2005; Ogle et al. 2008a, 2008b). Seeding guides may also cite climatic thresholds below which active seeding practices are not recommended (Anderson et al. 1957; Jordan, 1981). Unfortunately, the microclimatic requirements for germination, emergence, and seedling establishment are much more restrictive than the longer-term climatic requirements for maintenance of mature plant communities (Call and Roundy 1991; Peters 2000; Hardegree et al. 2003). Current state-and-transition models acknowledge that there are perhaps a limited set of potential trajectories for moving between undesirable and desirable vegetation states (Westoby et al. 1989; Batabyal and Godfrey 2002; Bestelmeyer et al. 2003; Briske et al. 2003, 2005, 2006, 2008; Bashari et al. 2008). Westoby et al. (1989) noted that many transition pathways between alternative states require the occurrence of a specific and perhaps infrequent series of climatic events.

> “virtually no direct experimental evidence exists to link specific range planting conservation practices to conservation effects *per se*.”

A Native sagebrush/bunchgrass plant community at Jacks Creek in southwestern Idaho (Photo: USFS) **B** Cheatgrass dominated site in southern Idaho after wildfire (Photo: USFS)

> “…successful establishment was frequently associated with average or above-average precipitation for either the entire year, or during the season of establishment.”

The range planting literature is somewhat biased relative to inferences that can be drawn from plant-material/climate interactions. Less than 60% of the studies reviewed for this synthesis reported weather conditions during the study, and less than half of these studies were replicated for year effects. In studies that reported weather conditions, successful establishment was almost always associated with average or above-average precipitation for either the entire year, or during the season of establishment. This implies that climatic thresholds exist below which management actions have little effect on establishment success. These thresholds may vary for species with different establishment requirements, but previous studies have not been designed to test this hypothesis specifically.

The strongest evidence for plant-material suitability for a given climatic region is derived from observation of historical relationships between species and climate, experience-based observation, and long-term assessment of persistence of planted species (Harris and Dobrowolski 1986; Shiflet 1994; Barbour and Billings 2000; Natural Resources Conservation Service 2006).

Plant-Material Development

NRCS has developed relatively specific and detailed recommendations for suitability of plant materials for different site conditions, climatic zones, and management objectives (Ogle et al. 2008a, 2008b). Plant-material recommendations for both native and introduced species are based primarily on plant-materials discovery, screening, and breeding programs by NRCS Plant Materials Centers, and other government research and agricultural experiment station programs (Hafenrichter 1948; Stewart 1950; Harlan 1951; Schwendiman 1956; Anderson et al. 1957; Schwendiman 1958; Harlan 1960; Roundy and Call 1988; Alderson and Sharp 1994; Asay et al. 2003; Erickson et al. 2004). Selected or bred plant materials deemed to have superior productivity, vigor, establishment, disease resistance and/or seed-production characteristics are then cultivated and released for development as commercial varieties (Schwendiman 1958; Johnson and Asay 1995; Asay et al. 2003). The more recent efforts in plant-material development and evaluation focus on selection for, or comparison of, specific ecological and physiological traits (Aguirre and Johnson 1991b; Johnson and Asay 1995; Arredondo et al. 1998; Jensen et al. 2005). These efforts incorporate and report more detailed experimental design information, but are often based on relatively controlled experimental conditions in the laboratory, greenhouse, or an agricultural field environment (Arredondo et al. 1998; Jones et al. 2003). The majority of current plant-material recommendations are based on evaluations of field performance that are not accessible through refereed journal publications (Stewart 1950; Schwendiman 1956; Great Plains Council 1966; Jensen et al. 2001; Lambert 2005; Ogle et al. 2008a, 2008b).

The literature documenting management-scale range planting is dominated by studies in which few inferences can be made about relative performance of different species and seed sources (Casler 1999). Very few studies are replicated in such a way that within- or between-species variability can be assessed (Kneebone and Cremer 1956; Pitman and Jaymes 1980; Asay and Johnson 1983b; Rumbaugh and Johnson 1986; Burner et al. 1988; Asay and Johnson 1990; Kitchen and Monsen 1994; Asay et al. 1996; Casler 1999; Asay et al. 2001; Vogel and Jensen 2001; Jones et al. 2003; Robins et al. 2007). About 60% of the rangeland planting studies surveyed for this synthesis evaluated performance of either a single seed lot, or a unique seed mix. In studies that evaluated more than one seed lot of a given species, only 6% were replicated at the seed-lot level.

Seed Quality

Seed-quality recommendations for range planting conservation practices are generally limited to those concerning germination testing, and the calculation of seeding rates based on estimates of pure live seed (PLS). Seed quality, however, has been evaluated in a number of studies that have correlated seed size and other morphological attributes to seedling emergence, growth rate, nutrient utilization, and seedling morphology and yield (Trupp and Carlson 1971; Carren et al. 1987a, 1987b; Limbach and Call 1995a, 1996; Smith et al. 2003).

Recurrent selection for increased seed size, deep-seeding emergence, or rapid seedling growth can result in plant materials with improved stand establishment. McKell (1972) and Kneebone (1972) recommended selection for seed mass to improve seedling vigor and establishment. McKell (1972) also emphasized the importance of rapid germination and pointed out that this is often a characteristic of weedy opportunistic grasses. Kneebone (1972) asserted that seed mass is highly heritable, and thus is a trait that will often be responsive to selection.

For cross-pollinated species, genetic manipulation through artificial selection or hybridization may be used to develop plant materials that increase the likelihood of seeding success. Kneebone (1972) suggested selection for high seed mass with hand screens, air columns, or gravity tables. This may be combined with selection for rapid germination under stress conditions, coleoptile length, and deep-seeding emergence. Large numbers of seeds and seedlings can be easily screened, a feature that lends itself to genetic improvement. This type of genetic manipulation is unsuitable for self-pollinated species where natural outcrossing and within-population genetic diversity is limited.

Seedbed Preparation and Planting Methods

The following conservation practice recommendations are directly or indirectly related to microclimatic management of the seed bed: surface soil modification, microsite improvement, seeding depth, seeding rate, timing of seeding, and weed control. Seedbed microsite improvement can consist of operations designed to reduce water loss and/or adverse thermal conditions in the seed zone by improving infiltration into the soil, improving water availability to the seed, reducing water loss to the atmosphere or reducing plant competition for water. This is accomplished through initial mechanical disturbance, soil firming and surface modification, control of seeding depth, application of soil surface amendments, and weed control (Roundy and Call 1988; Sheley et al. 1996).

Surface Modification

Soil surface modification is often justified by expectations of increased water availability to the seed, either by improving seed–soil contact, reducing the amount of surface area subject to evaporation, increasing infiltration and water-holding capacity, or by creating specific microsites that either receive or retain water more effectively (McGinnies 1959; Roundy et al. 1992). In some situations, cultivation without surface firming can increase the

A Seedbed disturbance caused by a rangeland drill (USFS, 2008) **B** Seedbed disturbance caused by a low-till drill (USFS, 2008) **C** Seedbed disturbance caused by a brillion cultipacker (Photo: Alex Boehm, 2009)

> Specific seedbed treatments to conserve water may not have much effect on establishment success in very wet or very dry years"

surface area subject to evaporation, reduce effective seed–soil contact, reduce seeding depth control, decrease hydraulic conductivity from deeper soil layers, and stimulate weed establishment if seeds are not effectively buried (McGinnies 1962; Kyle et al. 2007). Subsequent soil firming from press wheels or cultipackers improves hydraulic conductivity to the seed by reducing soil surface area and soil macroporosity (Hyder and Sneva 1956; McGinnies 1962). The bulk of range planting literature does not separate out treatment effects of soil-firming procedures, which are usually performed in conjunction with specific cultivation and planting procedures (Bement et al. 1965; McGinnies 1972; Slayback and Renney 1972). Studies that compare multiple seed-bed preparation methodologies often find differences in relative seeding success with different equipment and techniques, but specific inferences can only be made at the treatment level for a given site and year (Hubbard and Smoliak 1953; Hyder et al. 1955). Few studies of this type have been replicated adequately in multiple years or on multiple sites (Bement et al. 1965; Eckert and Evans 1967; Klomp and Hull 1972; Wood et al. 1982; Young et al. 1990; Bakker et al. 2003).

Animal trampling, land imprinting, pitting, furrowing, and rolling treatments have all been used in conjunction with broadcasting to capture or preserve moisture, and to press surface-applied seed into the soil (Hyder et al. 1955; Hyder and Sneva 1956; McGinnies 1959, 1962; Houston 1965; Haferkamp et al. 1987; Roundy et al. 1990; Winkel and Roundy 1991; Winkel et al. 1991a; Roundy et al. 1992; Ethridge et al. 1997). Animal ingestion and subsequent deposition of seeds in dung has also been used as a mechanism to disperse seeds into favorable microsites (Akbar et al. 1995; Andrews 1995; Ocumpaugh et al. 1996; Auman et al. 1998; Traba et al. 2003; Gokbulak and Call 2004). Differential establishment success relative to position of soil surface features has been reported, and is generally attributed to differences in microclimatic conditions (Anderson and Swanson 1949; Hyder and Sneva 1956; McGinnies 1959; Hull 1970; Bragg and Stephens 1979; Hauser 1982; Eckert et al. 1986; Roundy et al. 1992). Surface modification treatments have also been noted to push small seeds too far into the soil or to cause surface features to fill with soil from wind and water erosion, resulting in seed burial beyond establishment depth (Hyder and Sneva 1956; Kincaid and Williams 1966; McGinnies 1972; Slayback and Renney 1972; Winkel et al. 1991a). Positive effects of these surface features may be less relevant in very wet years when water is generally available, regardless of surface treatment, or in very dry years when plantings are unsuccessful regardless of seed-bed preparation technique (McGinnies 1968; Stuth and Dahl 1974; Wood et al. 1982; Eckert et al. 1986; Roundy et al. 1990; Winkel and Roundy 1991; Roundy et al. 1992; Romo and Grilz 2002).

Mulch Application

Application of mulch to improve range seeding success is frequently advocated as a mechanism to reduce water loss and moderate soil surface temperatures, although with the caveat that it is probably not cost effective for most rangeland applications (Lavin et al. 1981; McGinnies 1987; Ethridge et al. 1997). Relatively expensive soil surface amendments such as mulch are generally applied only after high-impact disturbance such as mine reclamation, or for mitigation of erosion after wildfire on topographically complex terrain (Jacoby 1969; Meyer et al. 1970; Lavin et al. 1981; Pinchak et al. 1985; Schuman et al. 1985; McGinnies 1987; Schuman et al. 1998; Whisenant 1999; Kruse et al. 2004; Groen and Woods 2008). An exception may be mulch production as a byproduct of mechanical shredding for control of juniper and other woody species (Brockway et al. 2002). Establishment of a cover crop to create standing-stubble mulch is usually limited to relatively small areas of major disturbance, or higher precipitation zones where grazing lands are being reclaimed from cultivation (Stroh and Sundberg 1971; Stubbendieck et al. 1973; Pinchak et al. 1985; Schuman et al. 1985; Hart and Dean 1986). Justification for mulching practices on rangelands is derived from greenhouse, laboratory, and modeling studies, all of which confirm general benefits of water conservation and mitigation of high temperature near the soil surface as a function of relative coverage (Hopkins 1954; Bond and Willis 1970; Chung and Horton 1987; Bristow and Abrecht 1989; Jalota 1993; Brar

and Unger 1994; Bussiere and Cellier 1994; Gill and Jalota 1996; Novak et al. 2000a, 2000b, 2000c; Giminez and Govers 2008), and field studies, most of which have been conducted after tillage or on severely disturbed, or otherwise extreme, sites (Dudeck et al. 1970; Meyer et al. 1970; Stubbendieck et al. 1973; Schuman et al. 1985; Hart and Dean 1986; Ethridge et al. 1997; Ji and Unger 2001; Dahiya et al. 2007; Groen and Woods 2008). Water conservation associated with mulch application on range seeding success may not be ecologically significant in very high or very low precipitation years or on some extreme rangeland sites (Gates 1962; Ludwig and McGinnies 1978; Lavin et al. 1981; Berg and Sims 1984; McGinnies 1987; Bristow 1988; Cione et al. 2002; Fulbright et al. 2006). For the 21 studies surveyed for this review that specifically evaluated mulch treatments, 62% concluded that mulch application improved establishment success. Regardless of the variable effects of mulch on seeding success, application of mulch for effective erosion control and soil stabilization is well documented (Meyer et al. 1970; Bautista et al. 1996; Fulbright et al. 2006; Groen and Woods 2008).

Seeding Depth

Successful germination and establishment is dependent upon placement of seeds in favorable soil microsites (Hyder et al. 1955; Harper et al. 1965; Young et al. 1990; Call and Roundy 1991; Winkel and Roundy 1991, Winkel et al. 1991b; Roundy et al. 1992; Chambers and MacMahon 1994; Sheley et al. 1996; Ott et al. 2003). A major assumption of many site-preparation treatments is that they increase the number of potential safe sites for germination and establishment either by covering the seed, by reducing soil water loss from around the seed, or by redistributing and concentrating resources (Anderson and Swanson 1949; Hubbard and Smoliak 1953).

Mechanical disturbance is generally necessary to incorporate seeds into the soil, thus reducing the risk of either desiccation or adverse thermal effects near the surface. Seeding depth recommendations from commonly cited seeding guides and technical references are relatively specific, but are based on rules of thumb regarding seeding depth as a function of seed size (Hull and Holmgren 1964; Plummer et al. 1968; Jordan 1981; Roundy and Call 1988; Jensen et al. 2001; Monsen and Stevens 2004; Lambert 2005; Ogle et al. 2008a, 2008b). The physical rationale for depth recommendations usually assumes a trade-off between increased water availability and increased energy requirements for emergence as a function of depth (Roundy and Call 1988; Call and Roundy 1991). In some cases, light or diurnal temperature fluctuation may regulate dormancy to ensure that the seeds germinate at an appropriate depth for a given species (Call and Roundy 1991; Ghersa et al. 1992; Traba et al. 2004). Seed predation has also been documented as a potential problem for surface-sown seeds (Nelson et al. 1970).

Evidence for depth effects is generally limited to studies conducted in a controlled environment, or over very small spatial scales in the field (Kinsinger 1962; Vogel 1963; Hull 1964). A major exception is for studies comparing the relative establishment of broadcast versus planted seeds. Of the 23 field studies surveyed for this review that specifically compared broadcast versus drill seeding, 73% concluded that drill seeding outperformed broadcast seeding. These studies, however, did not generally include quantification of

Brillion wheel for pressing broadcast seeds into the soil (Photo: USFS, 2006)

Standard 1X seeding rate of 247 seeds/m^2 or 23 seeds/ft^2 in a 0.25-m^2 frame (left). Middle frame shows a 2X seed rate and right frame shows a 5X seed rate (Photo: Alex Boehm, 2011)

the specific depth distribution after planting (Stewart 1950; Hyder et al. 1955; Douglas et al. 1960; Gomm 1964; Bement et al. 1965; Statler 1967; Shown et al. 1969; Nelson et al. 1970; McGinnies 1972; Drawe et al. 1975; Wood et al. 1982; Haferkamp et al. 1987; Ott et al. 2003). Relative seeding depth in field studies is often reported in the context of depth band settings on mechanical seeding equipment, but there are very few studies in which actual seeding depth has been quantified postplanting (Winkel and Roundy 1991; Winkel et al. 1991a, 1991b). Laboratory, greenhouse, and field comparisons of surface-sown versus planted seeds generally confirm that very small seeds establish more frequently from near-surface seed placement, larger seeds require soil cover for maximal performance, and seed performance drops dramatically below some threshold depth (Hull 1948; Stewart 1950; Douglas et al. 1960). Indian ricegrass (*Achnatherum hymenoides* [Roem. & Schulte.] Barkworth) has been extensively documented for its ability to germinate and emerge from relatively deep sowing depths, especially in sandy soils (Kinsinger 1962; Jones 1990; Young et al. 1994).

Broadcast and planting recommendations are generally not discretionary, as topographic complexity and economic considerations may preclude the use of planting equipment. Broadcast seeding rates are generally recommended at two to three times the rates for seed that can be incorporated into the soil (Stewart 1950; Hyder et al. 1955; Douglas et al. 1960; Gomm 1964; Bement et al. 1965; Statler 1967; Shown et al. 1969; Nelson et al. 1970; McGinnies 1972; Drawe et al. 1975; Wood et al. 1982; Haferkamp et al. 1987; Ott et al. 2003).

Seeding Rate

General seeding-rate recommendations from many technical sources appear to be based on a general standard for what could be considered a hypothetical dominant bunchgrass, planted at optimal depth, in a uniform, well-prepared, weed-free seed bed, in a favorable establishment year. The standard seeding rate for this hypothetical scenario seems to be roughly equal to a seed density of 1 million seeds per acre or approximately 23 seeds/ft^2 under historical, non-SI units of measure (Jordan 1981; Jensen et al. 2001; Monsen and Stevens 2004; Lambert 2005; Ogle et al. 2008a, 2008b). The most commonly recommended deviation from this hypothetical standard is to increase seeding rate by a factor of two–five for small seeds or for potential location-specific problems such as inadequate weed control, lack of site preparation, surface application of seeds, probability of drought, nonoptimal seeding season, or high levels of seed dormancy (Jordan 1981; Monsen and Stevens 2004; Thompson et al. 2006). Seeding-rate recommendations are also generally adjusted to reflect the total seed-mix ratio, and ideal expectations for composition of the desired mature plant community (Pyke and Archer 1991; Ogle et al. 2008a, 2008b). It is often difficult to assess numerical seeding rates, as the bulk of the literature reports rate in terms of weight of seed planted per unit land area. Weight-based recommendations in the technical literature, however, are generally supplemented by bulk seed density

information (Plummer et al. 1968; Jensen et al. 2001; Monsen and Stevens 2004; Lambert 2005; Ogle et al. 2008a, 2008b).

Seeding-rate recommendations are linked to microclimatic considerations, as increased seed numbers increase the probability of seeds reaching safe microsites, irrespective of active depth management (Harper et al. 1965; Call and Roundy 1991; Roundy et al. 1992; Chambers 1995). Relatively few studies reporting effects of seeding rate on establishment success are replicated in such a way to survey annual and seasonal variability in seed-bed microclimate (Schultz and Biswell 1952; Mueggler and Blaisdell 1955; Hull and Holmgren 1964; Launchbaugh and Owensby 1970; Hull 1972a, 1974b; Papanastasis and Biswell 1975; Vogel 1987; Masters 1997; McMurray et al. 1997; Williams et al. 2002). Some studies that include variable seeding rates were primarily designed to evaluate competition relative to weed-seed numbers, but in general, the literature supports the concept that higher seeding rates may enhance the likelihood of successful initial establishment (Vogel 1987; Sheley et al. 1999; Wiedemann and Cross 2000; Williams et al. 2002). Seeding-rate impacts remain highly dependent upon threshold requirements for water availability in the early stages of establishment, and individual seedling growth can be negatively impacted by both inter- and intraspecific competition later in development. The majority of the literature pertaining to seeding-rate effects is derived either from controlled environment and greenhouse studies, or field studies conducted in years where reported rainfall conditions were either average or above average (Francis and Pyke 1996; Sheley and Half 2006). Eiswerth and Shonkwiler (2006) evaluated a large number of range seeding sites and years in Nevada and determined that increased seeding rates led to higher seedling densities for nonnative grasses up to some maximum seeding rate. This study, however, did not analyze or report negative seeding results, and did not consider weather and climate conditions during the years that seeding occurred.

Planting Season

Most studies of planting-season effects on establishment success can be linked to climatic variability, and often to specific germination and dormancy syndromes of various seeded, nonseeded, and weedy species (Angevine and Chabot 1979). General planting-season recommendations require getting the seed planted in time to take advantage of the most favorable season for plant establishment (Hull 1948; Stoddart and Smith 1955; Plummer et al. 1968; McGinnies 1972; Vallentine 1979; Jordan, 1981; Roundy and Call 1988; Ries and Hofmann 1996; Monsen and Stevens 2004; Stevens 2004). In some cases, dormant-fall seeding is recommended well in advance of the optimal growing season to take advantage of all opportunities for potential establishment in a highly variable, and often arid or semiarid environment (Hull 1948; Stewart 1950; Douglas et al. 1960; Plummer et al. 1968; Young et al. 1969b; Nelson et al. 1970; Klomp and Hull 1972; Hart and Dean 1986; Young et al. 1994; Monsen and Stevens 2004). Dormant-fall seeding is also recommended when there are logistical concerns for use of mechanical equipment during wet-spring planting conditions, or to mitigate effects of unpredictable spring weather (Stewart 1950; Douglas et al. 1960; McGinnies 1973; Hart and Dean 1986). Seasonal timing of seeding may also be dependent on seasonality of weed competition and/or optimal timing of weed control measures (Bement et al. 1965; Robocker et al. 1965; Hull 1972a; Klomp and Hull 1972). The most favorable season for establishment varies regionally (Hatfield 1990): spring in Mediterranean-coastal and Intermountain West locations (Douglas et al. 1960; Nord et al. 1971; Hull 1972a; Harris and Dobrowolski 1986), summer monsoon in the southwestern desert (Jordan 1981; Abbott and Roundy 2003; Hereford et al. 2006), late spring through early summer in the Great Plains (Robertson and Box 1969; Hyder et al. 1971; McGinnies 1973; Hart and Dean 1986; Ries and Hofmann 1996; Frank et al. 1998; Romo and Grilz 2002), and late spring through early fall in some higher-elevation mountain sites (Hull 1966; Currie 1967; Lavin et al. 1973; Hull 1974a, 1974b). Postplanting microclimate must be favorable for growth, but also needs to remain favorable during the vulnerable period of seedling establishment (Hyder et al. 1971; McGinnies 1973; Frasier et al. 1987; Abbott and Roundy 2003). Eiswerth and Shonkwiler (2006) confirmed the relative

> “Of 52 studies surveyed for this review, all but two concluded that weed control was either necessary, or at least beneficial to successful establishment.”

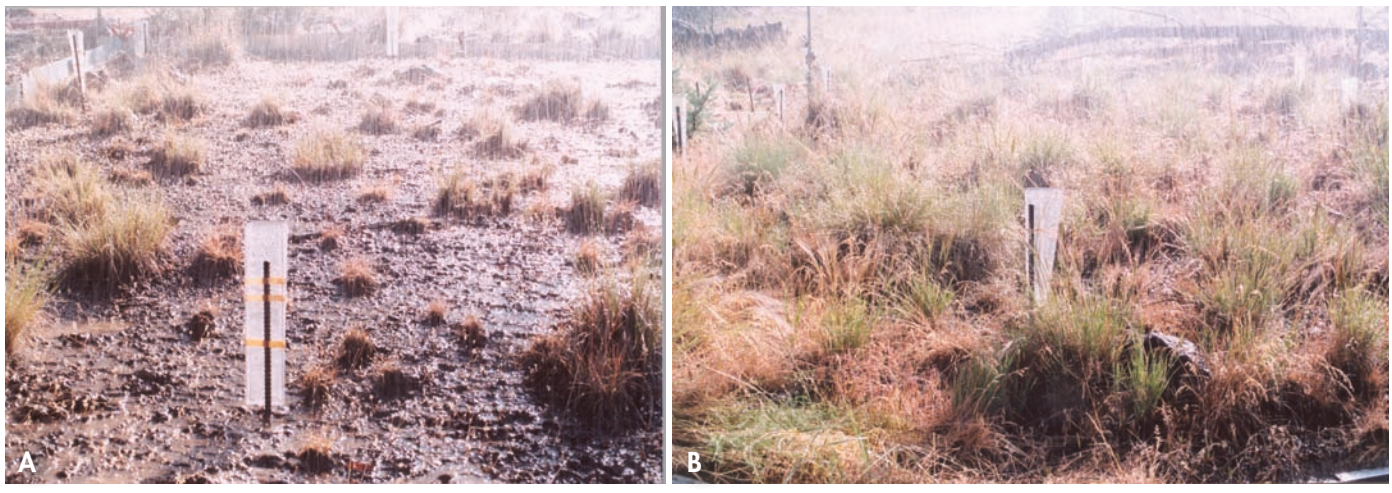
benefits of fall/winter-dormant seeding on intermountain rangelands in Nevada with the use of meta-analysis of long-term Bureau of Land Management fire-rehabilitation monitoring data. Very few experimental studies of seeding-season effects are replicated in more than 1 or 2 yr (Hull 1948; Douglas et al. 1960; Robocker et al. 1965; Hull 1974b; Ries and Hofmann 1996). Fall-dormant planting, however, was found to be superior to spring planting in 73% of Great Basin studies where planting season was evaluated.

Weed Control

Seed-bed preparation and planting method recommendations are designed to improve microclimatic conditions for desirable species, but also to reduce competition from undesirable plants (Lavin et al. 1973; Gonzalez and Dodd 1979; Ott et al. 2003; Mangold et al. 2007). Chemical or mechanical weed control, prior to the early stages of establishment, are generally required for establishment success of both native and nonnative plant species (Evans et al. 1970; Nelson et al. 1970; Klomp and Hull 1972; Stuth and Dahl 1974; Evans and Young 1978; Humphrey and Schupp 2002; Mangold et al. 2007). Of 52 studies surveyed for this review that included mechanical or chemical weed control, all but two concluded that weed control was either necessary, or at least beneficial to successful establishment. Efficacy of alternative mechanical and chemical weed control treatments is more extensively discussed elsewhere in this volume (Sheley et al., this volume).

NRCS TECHNICAL RECOMMENDATIONS AND THE RANGE PLANTING LITERATURE

General management recommendations, and associated NRCS technical references (e.g., Ogle et al. 2008a, 2008b), are consistent with current rangeland planting technical guidance and authorities (Vallentine 1979; Jordan 1981; Sours 1983; Redente and DePuit 1988; Roundy and Call 1988; Roundy 1996; Sheley et al. 1996; Whisenant 1999; Jensen et al. 2001; Monsen and Stevens 2004; Stevens 2004; Sheley et al. 2006). These recommendations and guidelines do not fundamentally differ from earlier-cited works that predate current standards for hypothesis testing, statistical inference, and experimental design norms (Stoddart and Smith 1943; Stewart 1950; Stoddart and Smith 1955; Anderson et al. 1957; Plummer et al. 1968; Vallentine 1979). Many of the historical references used to justify range planting practices, however, come from Agricultural Experiment Station reports, internal agency documents, and syntheses of unpublished field trials, as opposed to the refereed literature (McGinnies et al. 1963; Great Plains Council 1966; Plummer et al. 1968; Gomm, 1974; Cox et al. 1984; Call and Roundy 1991). With the exception of some specific plant-material selection and development programs, the underlying principles of these earlier recommendations were primarily based on previously established agricultural concepts, and a probabilistic assessment of best management practices derived from the practical experience and



A High erosion potential for soil surface with low plant cover caused by competition with western juniper. **B** Low erosion potential for soil surface with high plant cover after removal of woody-plant competition (Photo: Fred Pierson, 2003)

personal observations of land management professionals. The scientific literature from the more recent 40–50 yr has attempted to refine and to validate these commonly recommended practices experimentally. The more recent literature, however, is dominated by empirical studies that provide examples of field success for specific planting techniques, but, individually, are insufficiently replicated for general inferences (Call and Roundy 1991). We therefore surveyed 189 range planting field studies to draw the following general inferences regarding assumptions inherent to current seeding technical references, and range planting conservation practice documents:

- General recommendations supported by the aggregate literature must be prefaced by an acknowledgement that climatic conditions during the establishment year must be favorable. Ninety percent of the range planting papers surveyed report at least one successful treatment. Of the 57% that reported climatic conditions during the study, however, 89% claimed average or above-average precipitation in the year of establishment for the successful treatments. Over half of the studies that report successful establishment in a below-average precipitation year note that the seasonal distribution of precipitation was favorable during early seedling development.
- Few inferences can be made from the range planting literature about the relative establishment characteristics of alternative plant materials. Very few studies are designed and replicated in such a way that within- or between-species variability can be assessed. Over 35% of studies evaluated used only one seed lot or a unique seed mix, and 24% compared relative establishment among unique seed mixes. Of the 86 studies in which more than one seed lot of the same species were evaluated, only 6% were fully replicated, and 19% were partially replicated at the species level. Almost half of the studies used at least some named varieties, but only four specifically evaluated within-species variability. The strongest evidence for plant materials suitability is derived from observation of historical relationships between species and climate, experience-based observation, long-term assessment of persistence of planted species, and field trials conducted during the process of plant-materials selection and development.
- General conservation practice recommendations regarding site preparation and seeding methodology are generally supported from the aggregate literature. Drill-seeding treatments outperformed broadcast-seeding treatments in 73% of the studies that included a direct comparison. Application of mulch improved establishment success in 62% of the studies where there was a direct comparison. Increasing seeding rate was found to improve establishment success in 79% of the 24 studies where this was directly tested. Of the 52 studies that included mechanical or chemical weed control treatments, all but 2 concluded that weed control was either necessary or at least beneficial to successful establishment. Fall-dormant planting was determined to be superior to spring planting in 73% of the Great Basin studies where planting season was evaluated. Seed-bed preparation, seeding depth, planting season, and seeding-rate recommendations may be irrelevant in very dry and perhaps very wet years.
- The majority of range planting field studies are unreplicated in either space or time. Only 47% replicate planting years and 41% replicate site locations. The predominant form of treatment replication was within site, with 69% of studies having at least two, and 61% having at least three, within-site treatment replicates. Meta-analysis of studies that are individually underreplicated for general inferences is hampered by the fact that negative results are usually not published, and plant-materials selection is often based on *a priori* assumptions about their suitability for a given location.

ASSESSMENT OF SPECIFIC CONSERVATION EFFECTS

Evidence supporting positive conservation effects of alternative established plant communities is generally found in a separate body of literature than that examined in the Conservation Practices section of this

> “There is virtually no experimental evidence directly linking rangeland seeding to conservation effects.”

> Application of mulch to improve range planting success is probably not cost effective for most rangeland applications

chapter. The literature supports the concept that seeding, if successful, results in positive conservation effects. There is virtually no literature, however, directly linking rangeland seeding to conservation effects. We have limited our review to conservation effects related to water quality and erosion, water quantity, and soil carbon sequestration, because these conservation effects are not specifically addressed in the other chapters in this volume. Conservation effects related to livestock and wildlife needs, weed proliferation, and fire are left to the chapters in this volume concerning prescribed grazing, upland wildlife habitat management, herbaceous weed control, and prescribed burning.

Water Quality and Erosion

Very few studies directly link rangeland seeding to conservation benefits from improved water quality and reduced erosion (Wright et al. 1982; Brown et al. 1985; Beyers 2004). There is an extensive literature, however, documenting the relationship between rangeland soil cover and soil stability (Nearing et al. 2005; Bartley et al. 2006; Gimeno-Garcia et al. 2007). Removal of plant canopy cover by clipping may be insufficient to increase sediment loss in the short term when soil is still protected by basal vegetation cover and surface residues (Giordanengo et al. 2003; Gyssels et al. 2005; Nearing et al. 2005; De Baets et al. 2006). Range planting, *per se*, will not have a significant effect on soil stability unless sufficiently successful to provide adequate soil cover (Meeuwig 1965; Gifford 1970, 1972; Wright et al. 1982; Brown et al. 1985; Ziegler and Giambelluca 1998; Aguilera et al. 2003; Beyers 2004; Groen and Woods 2008). Short-term effects of site preparation, fire, or other treatments that precede range seeding or natural recovery, however, can significantly increase potential erosion in the near term (Williams et al. 1969; Gifford 1972, 1973; Tromble 1976; Roundy et al. 1978; Tromble 1980; Brown et al. 1985; Benavides-Solorio and Macdonald 2001; Gimeno-Garcia et al. 2007; Grismer 2007; Pierson et al. 2007). In most cases, the nature of this cover is less relevant than the issue of soil surface protection above some threshold level (Mergen et al. 2001; Aguilera et al. 2003; Descheemaeker et al. 2006). Some studies, however, have shown differential hydrologic effects under adjacent plant communities due to differences in growth and litter production, interception, water-use efficiency, or rooting depth and spread (Dunkerley 2002; Bhark and Small 2003; Kulmatiski et al. 2006). The relative impact of vegetation cover on erosion and runoff is also highly dependent on weather, slope and soil type (Aguilera et al. 2003; Bartley et al. 2006; Nichols 2006). Major soil loss after vegetation removal can be exacerbated by intense rainfall events (Gifford 1973; 1975; Garza and Blackburn 1985; Takar et al. 1990).

Vegetation affects soil stability and runoff water quality by protecting the soil from rainfall impact, increasing soil infiltration capacity, anchoring the soil mass, and preventing the development of rill erosion by slowing overland flow rates and increasing surface water flow paths (Tromble et al. 1974; Thurow et al. 1987; Pimentel and Kounang 1998; Aguilera et al. 2003; Imeson and Prinsen 2004; Puigdefabregas 2005). Invasive woody plants have been shown to suppress understory species to the point where insufficient soil surface cover can result in significant erosion, even under relatively low-intensity storm events (Davenport et al. 1998; Pierson et al. 2007; Petersen and Stringham 2008). Annual weed cover can affect seasonal patterns of evapotranspiration and soil water use (Kulmatiski et al. 2006; Prater and DeLucia 2006), but there is little evidence that they increase site runoff or erodibility when they are providing adequate soil cover (Singh 1969; Pierson et al. 2002; Wilcox and Thurow 2006; Pierson et al. 2007). In the Intermountain West, however, invasive annual weeds can increase the frequency of periods where vegetation cover can be reduced by wildfire (Brandt and Rickard 1994; Knapp 1996; Young and Longland 1996; Whisenant 1999). Indeed, the primary objective of most fire-rehabilitation seeding practices is to improve soil stability and reduce erosion (Richards et al. 1998; Bureau of Land Management 1999; Beyers 2004; Grismer 2007).

Application of mulch to improve range planting success is probably not cost effective for most rangeland applications (Lavin et al. 1981; McGinnies 1987; Ethridge et al. 1997). Mulch application, litter retention after site preparation, and mulch production as a result

of mechanical treatments for woody plant control, however, can have a direct conservation effect of reducing erosion and runoff on severely disturbed or highly erodible range sites, or on steep slopes (Meyer et al. 1970; Gifford 1975; Knight et al. 1983; Bautista et al. 1996; Brockway et al. 2002; Benik et al. 2003; Grismer and Hogan 2004, 2005; Fulbright et al. 2006; Groen and Woods 2008).

Water Quantity

Different plant materials and species may have different degrees of water-use efficiency and biomass production, but in arid rangeland systems, plants tend to use all available water in the soil profile in most years (Wright and Dobrenz 1973; Cable 1980; Trlica and Biondini 1990; Dugas and Mayeux 1991; Weltz and Blackburn 1995; Hester et al. 1997; Wilcox 2002; Huxman et al. 2005; Wilcox and Thurow 2006). Seeding grasses after shrub removal can result in increased stream flow in some Mediterranean-type climates, where the principal precipitation season is out of phase with the seasonal peak of evapotranspiration, or in systems where the woody plant material has access to groundwater (Hill and Rice 1963; Hibbert 1983; Wilcox 2002; Huxman et al. 2005; Huang et al. 2006; Wilcox and Thurow 2006). Shrubland conversion to grassland is not generally expected to result in an increase in water quantity in arid and semiarid upland rangeland systems, except where vegetation removal may increase overland flow directly to a stream channel (Gifford 1970; Wright et al. 1982; Bergkamp 1998; Wilcox 2002; Wilcox et al. 2003; Wilcox and Thurow 2006) or during extreme rainfall events where runoff may be affected by total plant cover (Weltz and Blackburn 1995; Quinton et al. 1997). The major exception to this would be the relatively large potential increase in overland flow after major vegetation disturbance, which may also generate unacceptable levels of soil erosion (Gifford 1973; Osborn and Simanton 1990; Takar et al. 1990; Johansen et al. 2001; O'dea and Guertin 2003; Pierson et al. 2007).

Carbon Sequestration

Svejcar et al. (2008) summarized the results of a 6-yr regional experiment that monitored seasonal carbon flux on western US rangelands and found that relatively good condition rangeland generally serves as a carbon sink, except in the driest areas of the desert southwest. The degree of carbon sequestration or loss varies primarily in response to seasonal and annual weather patterns (Conant et al. 2001; Flanagan et al. 2002; Jones and Donnelly 2004; Xu and Baldocchi 2004; Follet and Schuman 2005; Hastings et al. 2005; Derner and Schuman 2007; Svejcar et al. 2008). A change from tillage and annual cropping to perennial grass cover can greatly increase soil carbon-sequestration rates, but the effect is less on western rangeland soils than in more mesic areas (Conant et al. 2001; Guo and Gifford 2002; Sperow et al. 2003; Jones and Donnelly 2004; Derner and Schuman 2007). Relatively low sequestration rates, however, are offset by the relatively large land area occupied by rangelands (Scurlock and Hall 1998; Derner and Schuman 2007). Type conversion from woody plants to grasses can lower carbon-sequestration rates if the initial plant community has a higher net ecosystem production, but this would probably not be the case in most upland arid and semiarid rangeland systems (Huxman et al. 2003). Restoration of severely disturbed rangeland, and activities such as mine reclamation, can significantly improve carbon-sequestration rates (Follet and Schuman 2005; Derner and Schuman 2007).

Rooting depth, and effective water utilization for biomass production, can vary considerably among alternative vegetation types (Cline et al. 1977; Cable 1980; Yoder et al. 1998; Huxman et al. 2005; Seyfried and Wilcox 2006). As a significant portion of sequestered carbon can be deposited below ground by root growth (Scurlock and Hall 1998; Jones and Donnelly 2004; Rees et al. 2005), depth of rooting may be a consideration in selection of plant materials for rangeland seeding operations. Millions of acres of sagebrush–bunchgrass rangeland in the Intermountain West have been invaded by introduced annual weeds such as cheatgrass (*Bromus tectorum* L.). Cheatgrass-dominated systems are characterized by frequent recurrence of wildfire and are resistant to management actions that are designed to return them to a more desirable ecological state (Brandt and Rickard 1994; Knapp 1996; Young and Longland 1996). Carbon-sequestration rates can be nullified when vegetation is periodically removed by prescribed fire or wildfire (Suyker

> "Current conservation practice standards should include references to more ecologically based technical literature and specific guidance for both monitoring and adaptive management."

> The most useful potential technology for enhancing establishment success lies in development and utilization of relatively long-range weather-forecast technology"

and Verma 2001). Management activities, such as rangeland planting, that result in an increase in forage production can be expected to increase carbon-sequestration rates (Ma et al. 2000; Conant et al. 2001).

RECOMMENDATIONS

The aggregate literature generally supports both the existing conservation practice recommendations for rangeland seeding, and the inherent assumption that if these practices are successful, they will result in beneficial conservation effects. Current conservation practice recommendations, however, are relatively prescriptive in that they do not effectively address site- and year-specific variability, provide no mechanism for evaluating or adapting to unsuccessful or partially successful treatments, are oriented toward short-term management, and are not fully integrated with current, ecologically based models for management of alternative vegetation states. Current conservation practice standards should include references to more ecologically based technical literature and specific guidance for both monitoring and adaptive management. Additional research needs to be conducted to test the ecological underpinnings of existing and new ecological models for plant community dynamics explicitly, and to develop new tools to take advantage of existing and emerging knowledge of weather variability and anticipated shifts in regional climate. Individual seeding studies are seldom replicated sufficiently to make valid inferences in such a variable field environment. Therefore, monitoring protocols that can facilitate meta-analysis in support of more general inferences need to be developed.

KNOWLEDGE GAPS

Explicit Testing of New Conceptual Models for Dynamic Rangeland Systems

Call and Roundy (1991) recommended changes to the prevailing research approach to address problems inherent in highly variable rangeland systems more directly. A more general scientific understanding of vegetation change may now be achievable with the use of more recently developed conceptual models for understanding dynamic rangeland systems (Westoby et al. 1989; Bestelmeyer et al. 2003; Sheley et al. 2006). NRCS has already adopted some of these paradigms by utilizing state-and-transition-model concepts in the development of Ecological Site Descriptions, but these models do not currently form the basis for range planting conservation practice recommendations. These models integrate multiple processes and acknowledge multiple potential trajectories for plant community change and will require new and innovative approaches for validation and testing in the field.

Development and Utilization of Weather and Forecasting Tools

The stochastic nature of weather variability will require adoption of new concepts for evaluating revegetation and restoration success. Expectations for success need to be explicitly linked to the probability of favorable conditions for seed germination, emergence, and establishment (Krzysztofowicz 2001; Bakker et al. 2003). New technologies will need to be developed and utilized in order to use weather information to inform rangeland planting management decisions (Workman and Tanaka 1991; Peters 2000; Rayner et al. 2005; Andales et al. 2006).

The most useful potential technology for enhancing establishment success lies in development and utilization of relatively long-range weather-forecast technology specific to rangeland planting applications (Barnston et al. 1994, 2005; Garbrecht and Schneider 2007). Long-term weather forecasts in large portions of the Intermountain West are often merely synoptic descriptions of historical weather patterns and are not based on physical or empirical prediction of future weather conditions. It may be possible, however, to utilize historical weather and seeding data to construct models to assess the potential long-term benefits of adopting forecast/modeling technology in rangeland restoration planning (Batabyal and Godfrey 2002; Schneider and Garbrecht 2003, 2006; Bashari et al. 2008). Similar technology is in relatively common use for more traditional agricultural applications and for some rangeland applications (Schneider and Garbrecht 2003; Doblas-Reyes et al. 2006; Schneider and Garbrecht 2006; Baigorria et al. 2008; O'Lenic et al. 2008).

Monitoring native planting at the site of the Scooby fire in northern Utah (Photo: USFS, 2010)

Even low-resolution weather forecasts would increase the probability of successful native plant establishment if seeding decisions in the fall could be based on the anticipation of favorable conditions of seedbed microclimate in the subsequent winter and spring (Hardegree et al. 2003; Hardegree and Van Vactor 2004). Weather forecasts could be used to initiate contingency plans in areas that have been previously identified for restoration, and for which premanagement logistics of equipment, personnel, and plant materials are in place (Bakker et al. 2003; Westoby et al. 1989). Separation of restoration planning objectives from the wildfire cycle would also simplify the problem of predicting management needs for native germplasm (Richards et al. 1998). Historical climate records could provide a relatively stable estimate of the probability of favorable establishment years that could be used to predict acquisition and storage requirements for native seed over the long term.

Biodiversity and restoration planning objectives may require multiple-year strategies for replacement of nonnative species only after initial site stabilization and suppression of annual weed competition (Bakker et al. 2003; Cox and Anderson 2004). Weather and climatic limitations require definition of realistic goals when establishing rehabilitation and restoration planning objectives (Call and Roundy 1991; Hobbs and Norton 1996; Ehrenfeld 2000; Jones 2003). Asay et al. (2001) argue that the relatively harsh climatic conditions on many rangelands may preclude the realistic use of many native plant materials in favor of adapted nonnative species. In some years, and on some sites, it may be prudent to plant more easily established nonnative species, particularly after wildfire or other disturbance, when the principal objective of rangeland planting may be soil stabilization. Biodiversity and restoration objectives could then be addressed in years when climatic conditions are amenable (Holmgren

“Adoption of minimum experimental design standards would facilitate meta-analysis of future range planting studies.”

and Scheffer 2001; Bakker et al. 2003; Hardegree et al. 2003; Cox and Anderson 2004; Hardegree and Van Vactor 2004).

Plant-Materials Program Development and Testing

Previous plant-materials development has focused on productivity, vigor, establishment, disease resistance, seed production, and specific ecological and physiological traits deemed to confer superior performance or adaptation. Establishment, persistence, and invasion resistance of seeded plant communities may be enhanced by identification and selection of plant materials with functional traits similar to the various highly competitive invasive species (Arredondo et al. 1998; Pokorny et al. 2005; Funk et al. 2008). Functional traits common to many weedy invaders include high relative growth rate, specific leaf area, leaf nitrogen content, and resource-use efficiency (Aguirre and Johnson 1991a, 1991b; Grotkopp et al. 2002; Pokorny et al. 2005; Grotkopp and Rejmanek 2007; James and Drenovsky 2007; Funk et al. 2008).

Development of herbicide-resistant native grass plant materials may be a useful area of future research. In recent years, interest has increased in using acetolactate synthase (ALS) inhibitors such as imazapic to reduce annual grass competition with desirable perennials. Imazapic damages fast-growing tissues, especially meristems, so weedy annual grasses and crucifers are controlled with less damage to perennials (Shaner and O’Conner 1991). Some residual activity is present, so annual grass control is still achieved a year following application (Davison and Smith 2007). An advantage of this herbicide is that many desirable nongrass species, particularly legumes and composites, are relatively resistant. Development of ALS-inhibitor resistance has been quite successful in several crop species (Tranel and Wright 2002); thus development of native plant materials with such resistance is likely to be successful. When such materials are developed by traditional plant-breeding methodologies, no special Environmental Protection Agency clearance is required prior to release.

For many years, the NRCS and Agricultural Research Service have routinely evaluated released and experimental materials as part of their ongoing plant-material research and development programs. For the rangelands

Wildfire disturbance at the site of the Crowbar fire in south-western Idaho (Photo: USFS, 2010)

of the semiarid west, these trials are typically dormant planted in late fall for spring emergence. Comparisons between plant species and among plant materials within species are made in order to characterize released plant materials and to justify the release of new plant materials. The single most important trait of these trials is seedling establishment. These trials are typically replicated complete block design experiments subject to statistical analysis, but are usually analyzed as individual experiments rather than as a collective whole. Because these trials are conducted every year, a large volume of data have been collected. More robust comparisons of species and plant materials could be made if these data were compiled and subjected to meta-analysis. A Web-accessible program to update and combine data sets for analysis would greatly increase the statistical power of plant-material evaluations and improve decision making. With the use of additive main and multiplicative interaction (AMMI) statistical models, NRCS PRISM climatic data could be included in the data set to facilitate recommendations that take into consideration environmental parameters.

Direct Conservation Effects of Plant Materials on Nutrient Cycling

Ecosystem disruption commonly results in a shift in nutrient-cycling dynamics from systems where nutrients, such as carbon (C) and nitrogen (N), are quickly sequestered by plants and microorganisms to systems that contain relatively greater amounts of available nutrients (Norton et al. 2007). These systems are more susceptible to weed dominance, leach more mineral N, and sequester less carbon than effectively functioning systems. Although these systems have higher N mineralization rates, soil N concentration is lower (Kulmatiski and Beard 2006). Nutrient-cycling conservation effects could be cited as an additional positive purpose for this conservation practice standard.

Adoption of Standard Protocols for Evaluating Success and Development of Meta-Analysis of Field Trials

The majority of range planting studies do not measure critical environmental factors affecting success, but only measure relative treatment effects (Call and Roundy 1991; Vargas et al. 2001). Range planting studies also tend to extrapolate results obtained from atypical sites and conditions over larger areas (Cox et al. 1984), and are seldom replicated in multiple seeding years (Casler 1999). Generally, high variability in experimental procedures often produces unique individual studies from a complex combination of unique site preparation, plant materials, seeding rate, soil conditions, and weather during only 1 or 2 establishment years. An important recommendation is adoption of minimum experimental design requirements for publication of range planting studies relative to specific inferences that are of principal interest (Casler 1999; Vargas et al. 2001).

Bluebunch wheatgrass seedlings (Photo: Lori Ziegenhagen, 2007)

Success metrics are highly variable and often consist of relative ranking of treatment effects, and there has been very little research to evaluate alternative criteria for quantification of success (Ries and Svejcar 1991). The majority of range planting studies only consider treatment effects in the first year after planting, and studies that are monitored for longer periods are generally not replicated for planting-year effects (Casler 1999). Many studies that have monitored range planting results in the very long term have noted significant changes from what would have been measured only 1–3 years postplanting (Bleak et al. 1965; Hull 1971a, 1973; Lavin and

Standard rangeland drill conducting a native seeding following the East Humboldt fire, Elko Co., NV. (Photo: USFS, 2008)

Johnson 1977; Eck and Sims 1984; Harris and Dobrowolski 1986).

In general, most individual studies within the range planting literature are insufficiently replicated to extract valid inferences about weather and climate effects, site effects, plant-materials effects, and seeding rate. The dominant level for validation in the currently available literature derives from interspersion and within-location replication of seed-bed preparation treatments. These studies, and a large amount of data contained in conference proceedings, technical reports, and internal-agency documents, might be subject to valuable meta-analysis of treatment effects that are difficult or impossible to replicate in the context of a stand-alone journal publication (Durlak and Lipsay 1991; Gurevitch et al. 1992; Adams et al. 1997; Michener 1997; Gurevitch and Hedges 1999; Osenberg et al. 1999a, 1999b; Gurevitch et al. 2001; Johnson 2006). Much of this information may only be suitable for low-level meta-analysis similar to the summary statistics used here to document gross treatment effects. It may be possible, however, to develop guidelines for establishing some common experimental design features for future studies that may be amenable to more sophisticated meta-analysis.

Another underutilized research resource is the incorporation of extensive management-level monitoring information into a scientific database format (Pastorok et al. 1997). Eiswerth and Shonkwiler (2006) used a Bureau of Land Management data set to evaluate postfire management treatment effects on seeded nonnative grasses, sagebrush, and annual weeds as a function of range site, soil type, and seeding prescription. Unfortunately, this data set did not evaluate impacts of weather and climate variability. Effective utilization of these types of data may also require some degree of coordination within and between management agencies to adopt similar monitoring protocols. NRCS Conservation Practice Standards could be improved by establishing standard monitoring requirements to assess both the effectiveness of specific management recommendations and conservation effects of successful practices. Monitoring requirements, however, should be based on an explicit experimental design that would facilitate future meta-analysis.

CONCLUSIONS

There is virtually no refereed journal literature directly linking NRCS rangeland seeding conservation practices to specific conservation effects. The aggregate literature, however, generally supports general conservation practice recommendations for rangeland seeding, and the potential conservation benefits should these practices result in successful establishment of a more desirable plant community. A major limitation to current conservation practice recommendations is that they do not explicitly acknowledge or provide management guidance to deal with the high variability in soil microclimate during germination, emergence, and early seedling development. Additional guidance is warranted to provide recommendations for monitoring and adaptive management in these arid and semiarid rangelands. Future research efforts would also benefit from experimental designs that were amenable to meta-analysis, as inferences from individual rangeland planting trials are generally limited to specific site conditions and plant materials.

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CHAPTER

5

A Scientific Assessment of the Effectiveness of Riparian Management Practices

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“The major objective of riparian habitat conservation practices is to effectively manage riparian vegetation, stream channel, and soil resources to protect or enhance these ecosystem services”

A Scientific Assessment of the Effectiveness of Riparian Management Practices

5

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INTRODUCTION

This chapter evaluates the ecological effectiveness of the major purposes and expected benefits of 21 riparian management practices as described in the US Department of Agriculture, Natural Resources Conservation Service (USDA NRCS), National Conservation Practice Guidelines (Table 1). The ecological benefits described in the standards for these practices include the following:

- Wildlife habitat
- Water quantity and quality
- Stream bank and soil stability
- Carbon storage
- Plant and animal diversity

Riparian management encompasses many activities and practices that are applied directly to the riparian zone or that are applied in the uplands to influence the riparian zone. To meet numerous riparian management goals, conservation practices are often applied as a suite of practices called a resource management system. A resource management system may include several practices (e.g., prescribed grazing, stream crossing, riparian herbaceous cover) selected to meet site-specific conditions and objectives. Riparian areas occur along watercourses or near water bodies. They are different from surrounding lands because of unique hydrologic, soil, and plant characteristics that support important ecosystem functions and services.

Riparian areas occupy the transitional area between the terrestrial (dry) and aquatic (wet) ecosystems. Rangeland riparian areas include the stream, stream channel, and adjacent riparian vegetation. These areas also include seeps, springs, and small wetlands that have greater soil water relative to surrounding uplands. This does not include marshes, impoundments, estuaries, and other wetland habitats. Although riparian areas constitute only a fraction of the total land area on western rangelands, they generally support greater overall plant and animal species diversity, richness, and productivity than adjacent uplands. Access to riparian areas in rangeland systems is usually critical to sustaining the productive potential of the surrounding landscape. Riparian areas are often relatively long and narrow in relation to other landscape features. This characteristic creates significant interaction with other ecological sites within the landscape, supporting the exchange of materials and energy within the landscape.

Riparian corridor in eastern Oregon. (Photo: Chad Boyd)

Numerous studies in the western United States have shown that riparian areas have been negatively impacted by timber harvest, road building, irrigation, grazing, and other human activities (Kauffman and Krueger 1984; Fleischner 1994; Magilligan and McDowell 1997; Belsky et al. 1999). In many cases, these systems have been altered (e.g., down-cutting, head-cutting, and stream bank alteration) to the point that past geomorphic structure and function cannot be restored and returned to former conditions. Additionally, installation of dams and diversion of water have altered runoff timing and amounts, often resulting in irreversible changes in riparian characteristics. Where irreversible changes have occurred, some new desired condition becomes the objective of restoration.

Because grazing is such a widespread practice on public and privately owned rangelands, assessment of grazing management practices is a significant part of this review. Platts (1978, 1990) rated the effect of several grazing strategies for stream–riparian habitat values based on his observations and professional

TABLE 1. List of 20 riparian practices and their expected ecosystem services.

Practice name	Code	Ecosystem services				
		Wildlife habitat	Water quality and quantity	Stable stream banks and soils	Carbon storage	Diverse plant and animal communities
Animal trails and walkways (feet)	575		X	X		
Brush management (acres)	314	X	X	X		X
Channel bank vegetation (acres)	322	X	X	X		
Conservation cover (acres)	327	X	X	X		
Critical area planting (acres)	342			X		
Fence (feet)	382	X	X	X		X
Filter strip (acres)	393		X			
Pest management (acres)	595	X	X			
Prescribed burning (acres)	338	X				X
Prescribed grazing (acres)	528	X	X	X		X
Range planting (acres)	550	X	X	X	X	X
Riparian forest buffer (acres)	391	X	X	X	X	X
Riparian herbaceous cover (acres)	390	X	X	X	X	X
Stream crossing	578		X	X		
Stream habitat improvement and management (acres)	395	X		X		
Stream bank and shoreline protection (feet)	580	X		X		
Tree/shrub establishment (acres)	612	X	X	X	X	X
Upland wildlife habitat management (acres)	645	X				
Use exclusion (acres)	472		X	X		X
Watering facility (no.)	614		X	X		

experience (Table 2). Similarly, Kovalchik and Elmore (1991) rated the compatibility of grazing systems with willow-dominated communities (Table 3). While the effects of many of these grazing systems on riparian areas have been documented in case histories, rarely have they been tested with rigorous experimental designs and appropriate statistical analyses (Larsen et al. 1998). Both of these evaluations indicate that continuous grazing is not compatible with riparian areas and that rest or deferment from grazing, inherent in various forms of rotational grazing, tend to improve the riparian habitat values addressed in Tables 2 and 3. Continuous grazing often results in heavy grazing use of the riparian area because livestock are attracted to riparian areas from the adjacent uplands. Even if the pasture is lightly stocked, grazing may be heavy because livestock preferentially use the riparian zone. Improperly applied rotational grazing systems can also result in heavy grazing and damage to riparian habitat.

The objective of livestock grazing strategies and practices has been to increase plant and litter cover, encourage growth of desirable plant species, improve plant species composition, increase plant vigor, and protect riparian soil and stream banks from erosion. Grazing tactics or practices for maintaining

or rehabilitating riparian areas include 1) controlling the timing and duration of riparian grazing by fencing riparian pastures within existing pastures, 2) fencing riparian areas to exclude livestock from riparian areas, 3) changing the kind and class of livestock, 4) reducing duration of grazing, 5) reducing grazing intensity, and 6) controlling season of use (Clary and Webster 1989; Platts and Nelson 1989). Annual management objectives for vegetation attributes (e.g., herbaceous plant stubble height, woody plant utilization, and vegetative ground cover) are frequently recommended or required to guide year-to-year grazing management decisions (Bauer and Burton 1993; Hall and Bryant 1995; Clary and Leininger 2000). The assumption is that meeting annual management objectives will be compatible with long-term resource objectives (e.g., stream bank stability, recruitment of woody plants, clean water; Clary and Leininger 2000).

Scientific documentation that livestock grazing could damage riparian areas began in the 1980s (Skovlin 1984) and has been documented in numerous symposia (e.g., Warner and Hendrix 1984; Johnson et al. 1985; Gresswell et al. 1989; Meehan 1991; Clary et al. 1992), literature reviews (Platts 1981, 1982, 1991; Kauffman and Krueger 1984; Skovlin 1984; Chaney et al. 1990, 1993; Armour et al. 1994; Fleischner 1994; Rhodes et al. 1994; Kattelmann and Embury 1996; Ohmart 1996), and government reports (US Department of the Interior, Bureau of Land Management [USDI BLM] 1994; US General Accounting Office [US GAO] 1988). Recognizing that riparian ecosystem services need to be protected, USDA NRCS, along with many other federal and state resources agencies, began to apply existing conservation practices and to implement new practices with the goal of protecting and improving riparian habitats.

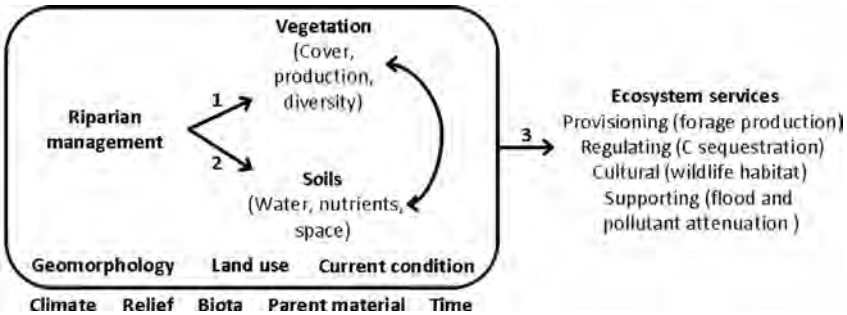
DESCRIPTION OF CONSERVATION PRACTICES AND BENEFITS

More than 40 practices in the USDA NRCS National Conservation Practice Guidelines (USDA NRCS 2003) were identified as having potential for application to riparian ecosystems. For this review, we narrowed the practices in Appendix I to a shorter list of 20 that are often associated with rangeland or pasture systems (Table 1). The *purposes or anticipated benefits* stated in the practice standards for these 20 practices can be summarized into five main ecosystem services: 1) high-quality and abundant fish and wildlife habitat, 2) clean and plentiful water supply, 3) stable stream banks and riparian soils supporting hydrologic functions such as flood and pollutant attenuation, 4) carbon sequestration, and 5) diverse, rich, productive plant and animal communities (Table 1). However, the major objective of riparian habitat conservation practices is to effectively manage riparian vegetation, stream channel, and soil resources to protect or enhance these ecosystem services (Fig. 1).

Objective and Approach

Recognizing that anticipated benefits of management practices applied to riparian habitats are mediated by resource availability, especially water, we developed a conceptual model that links management practices to vegetation attributes and resource constraints (Fig. 1). The model acknowledges the overriding importance of state factors such as climate, parent material, relief, geomorphology, past and contemporary land uses, and disturbances at the watershed and

FIGURE 1. Conceptual model of the effect of riparian conservation practices on vegetation, soils, stream banks, and ecosystem services. The outcome of conservation practices are contingent on the biophysical context, which is set at a coarse level by state factors such as climate, relief, parent material, and the age of the riparian area. These state factors interact to shape the geomorphology and current condition of the riparian zone. Land use at the landscape level of organization, including conservation practices in uplands, is considered a state factor because it is controlled by humans. Within the bounds set by a particular combination of state factors, riparian management can affect vegetation and soils. In addition to these direct effects, feedbacks exist between vegetation and soils that condition their individual and collective response to management. Ecosystem services in the form of forage production, carbon sequestration, wildlife habitat, and flood and pollutant attenuation emerge from these dynamics processes.



Practices such as fencing to manage grazing pressure on riparian vegetation and soils are anticipated to enhance riparian-based ecosystem services. (Photo: Ken Tate)

larger spatial scales on riparian vegetation and soils. It is widely documented that single or cumulative watershed-scale management practices (e.g., upland brush management, upland grazing management) and disturbances (e.g., fire, road construction) can affect riparian area functions, services, and response to site-specific management practices. For the purposes of this review, we focused on the interaction of various management practices with riparian soil and stream bank resource availability (i.e., water, nutrients, oxygen), and vegetation. We used the model illustrated in Figure 1 to generate 21 hypotheses that could be evaluated using published experimental data. The experimental data associated with these selected practices was identified by reviewing primarily peer-reviewed literature. Support for most hypotheses is summarized and incorporated into appendices to provide an evidence-based assessment of the effectiveness of these riparian management practices.

EVALUATION OF RIPARIAN MANAGEMENT PRACTICES

We classified 21 hypotheses into three riparian management purposes: 1) protection or restoration of vegetation attributes, 2) protection or restoration of stream channel and riparian soil stability, and 3) direct or indirect protection or improvement of ecosystem services (Fig. 1). In this section, each of the 21 hypotheses is evaluated against the supporting experimental data.

Practices That Protect or Restore Vegetation Attributes

Hypothesis 1: Management of Time, Intensity, Season, and Duration of Grazing Affect Herbaceous Species Composition.

Grazing systems facilitate control of season of use, frequency of use, duration of use, grazing intensity, and livestock distribution. However, herbaceous plant community response to grazing management is often difficult to predict because the responses are contingent on resource availability, namely, water (Stringham et al. 2001; Poole et al. 2006). Resource availability is moderated by the biophysical characteristics of the riparian area and its watershed (Fig. 1; Goodwin et al. 2008). In general, increasing grazing intensity results in a reduction of slower-growing, larger-seeded plant species (i.e., competitive species sensu Grime 1979) that are often considered desirable, depending on management objectives. As grazing intensity increases, the abundance of faster-growing, small-seeded species (i.e., ruderals) increases; however, this response may be less prominent where water is in abundant supply. Moreover, where water is limiting or the supply is erratic, ruderals may dominate even with little or no grazing. This model is an oversimplification because in many riparian systems, resource limitations (e.g., moisture, nutrients, and temperature) occur in transient pulses (Seastedt and Knapp 1993). Flooding events may create microsites where only species tolerant of anoxia can persist.

Lucas et al. (2004), working in New Mexico, found little effect of grazing intensity (no, low, and moderate) on herbaceous structure (cover, biomass) and composition (diversity). However, cool-season grazing promoted herbaceous diversity over warm-season grazing. The authors were adamant that grazing management affects streams in site-specific ways; hence, no single prescription is warranted for riparian management. This echoes findings of Jackson and Allen-Diaz (2006), who found highly variable interannual community characteristics in spring-fed wetlands that appeared unrelated to grazing intensity, while subsequent first-order streamside vegetation appeared directly linked to grazing treatments.

Lunt et al. (2007), working in a southeastern Australian riparian forest, showed that grazing exclusion had minimal impacts on understory composition and structure over a 12-yr period, attributing this to the fact that their system was nonequilibrial and responded more to abiotic factors than to biotic factors, such as grazing management. Clary (1999) found that all grazing treatments (0, 20–25%, and 35–50% utilization) resulted in increased plant species richness on streamsides and meadows as the systems recovered from historic heavy grazing. This indicates that the ecological condition of the riparian habitat at the onset of the study has important implications for the potential outcomes that may result from various management practices.

Kauffman et al. (1983a) observed a phenological shift in the herbaceous plant community of mesic and hydric riparian zones in eastern Oregon that they ascribed to quicker drying of grazed soils resulting from greater solar insolation incident on the soil surface. Their data showed an increase in undesirable plant species with grazing compared to exclosures, though the experimental design was weak and no estimate of uncertainty was reported. The grazing prescription during this study was 75% utilization of bluegrass (*Poa* spp.) meadows.

Lyons et al. (2000a) focused on the effects of different types of riparian vegetation on small streams in central North America and indicated that without grazing, these zones will become dominated by woody species that reduce stream bank stability. Paine and Ribic (2002) found more diverse plant communities and wildlife habitat when grassy buffer strips were present along riparian zones compared to woody-dominated riparian zones. In contrast, Carline and Walsh (2007) show that in Pennsylvania, exclusion of grazing for 3–5 yr, from formerly heavily stocked pastures, resulted in vegetation cover increases from 50% to 100%.

Our review of 11 reports found substantial support for the hypothesis that grazing intensity influences herbaceous species composition. However, managers should be aware that grazing effects on species composition may be influenced by the availability of resources, such as water and nutrients. Three of these studies (Lucas et al. 2004; Jackson and Allen-Diaz 2006; Lunt et

> “Eleven studies supported the hypothesis that grazing intensity influences herbaceous species composition however, grazing effects on species composition may be influenced by the availability of resources, such as water and nutrients”

TABLE 2. Evaluation and rating of grazing strategies for stream–riparian-related fisheries values based on observations of Platts (1990).

Strategy	Level to which riparian vegetation is commonly used	Control of animal distribution (allotment)	Stream bank stability	Brushy species condition	Seasonal plant regrowth	Stream–riparian rehabitative potential	Rating
Continuous season-long (cattle)	Heavy	Poor	Poor	Poor	Poor	Poor	1[1]
Holding (sheep or cattle)	Heavy	Excellent	Poor	Poor	Fair	Poor	1
Short duration–high intensity (cattle)	Heavy	Excellent	Poor	Poor	Poor	Poor	1
Three herd–four pasture (cattle)	Heavy to moderate	Good	Poor	Poor	Poor	Poor	2
Holistic (cattle or sheep)	Heavy to light	Good	Poor to good	Poor	Good	Poor to excellent	2–9
Deferred (cattle)	Moderate to heavy	Fair	Poor	Poor	Fair	Fair	3
Seasonal suitability (cattle)	Heavy	Good	Poor	Poor	Fair	Fair	3
Deferred rotation (cattle)	Heavy to moderate	Good	Fair	Fair	Fair	Fair	4
Stuttered deferred rotation (cattle)	Heavy to moderate	Good	Fair	Fair	Fair	Fair	4
Winter (sheep or cattle)	Moderate to heavy	Fair	Good	Fair	Fair to good	Good	5
Rest–rotation (cattle)	Heavy to moderate	Good	Fair to good	Fair	Fair to good	Fair	5
Double rest–rotation (cattle)	Moderate	Good	Good	Fair	Good	Good	6
Seasonal riparian preference (cattle or sheep)	Moderate to light	Good	Good	Good	Fair	Fair	6
Riparian pasture (cattle or sheep)	As prescribed	Good	Good	Good	Good	Good	8
Corridor fencing (cattle or sheep)	None	Excellent	Good to excellent	Excellent	Good to excellent	Excellent	9
Rest–rotation with seasonal preference (sheep)	Light	Good	Good to excellent	Good to excellent	Good	Excellent	9
Rest or closure (cattle or sheep)	None	Excellent	Excellent	Excellent	Excellent	Excellent	10

[1]Rating scale based on 1 (poorly compatible) to 10 (highly compatible) with fishery needs.

al. 2007), which were conducted in semiarid rangeland, concluded that grazing intensity effects were either nonexistent or overwhelmed by abiotic drivers. There was some support for a season, duration, or frequency of grazing effect on herbaceous species composition, but these studies were conducted mainly in the mesic upper Midwest or eastern grasslands (Lyons 2000b; Carline and Walsh 2007), where resources such as water and nutrients are typically in greater and more consistent supply. Studies that focused on mesic systems supported an increase in woody species with grazing exclusion.

Hypothesis 2: Management of Time, Intensity, Season, and Duration of Grazing Can Influence Aboveground Herbaceous Productivity. Compensatory growth is the stimulation of net primary productivity (NPP)

by defoliation such that regrowth compensates for the biomass removed by the defoliation process (Bartolome 1993). Overcompensation occurs when defoliation results in production that exceeds that of undefoliated plants. Riparian herbaceous vegetation is often very productive since riparian areas usually possess abundant nutrients and water. Hence, it is plausible that grazing management that does not degrade resource availability may also result in compensatory growth, as was found by Boyd and Svejcar (2004) in eastern Oregon. This mechanism was implicated by Jackson et al. (2006) as the reason for greater nitrate loss from spring-fed wetlands of the Sierra Nevada foothill oak woodlands. In this study, grazing stimulated production compared to no grazing, which promoted uptake of nitrate entering the wetlands from the surrounding landscape. Alternatively, removal of grazing resulted in an immediate increase in total standing biomass, but this biomass accumulated, as dead plant material, on the surface and suppressed subsequent productivity, a phenomenon also observed by Popolizio et al. (1994) in a Colorado riparian zone.

Huber et al. (1995) observed lower standing biomass under moderate grazing intensity compared to low and no grazing treatments, which were not different from each other. Caution must be used when interpreting peak standing biomass data because it is difficult to know whether a response to the treatment or the treatment itself is being measured. Productivity could be equal to or greater than the control, but standing biomass may be lower because of livestock utilization.

Studying grazing effects of pack stock in Sierra Nevada mountain meadows, Cole et al. (2004) found reduced vegetation productivity over 5 yr in three meadow communities. Stohlgren et al. (1989) conducted a clipping experiment on high-elevation subalpine meadows of the Sierra Nevada. They found that clipping for 5 yr to simulate heavy grazing negatively affected productivity in wet and mesic meadows but not in dry *Carex exserta* meadows. The authors caution that these results cannot be extrapolated to address grazing at light or moderate levels. However, these results support the general notion that grazing effects on productivity are likely to be more pronounced in systems where resource availability is relatively high, such as mesic compared to dry meadows, where environmentally driven resource limitation has a stronger influence. This is to say not that grazing management has no effect in resource-poor systems but rather that productivity is inherently low or variable and therefore less coupled to management.

Late-season clipping in a Sierra Nevada mountain meadow had no consistent effects on above- and belowground response variables, such as root growth and photosynthetic rates (Martin and Chambers 2002), similar to the late-season clipping results of Clary (1995, 1999). Kluse and Allen-Diaz (2005) clipped Sierra Nevada meadows dominated by *Deschampsia caespitosa* and *Poa pratensis* early in the growing season and found reduced productivity in both species but no shift in relative species abundance. Huber et al. (1995) found that light grazing of a Sierra Nevada meadow resulted in vegetation biomass similar to ungrazed meadows but encouraged cattle to graze away from streamside edge compared to heavy grazing. Allen and Marlow (1994) found that beaked sedge (*Carex rostrata*) tolerated light to moderate grazing in early summer and fall if there was at least 60 d of rest between grazing periods to allow production of new photosynthetic tissue.

Nine peer-reviewed reports support that grazing intensity can influence herbaceous productivity. Two reports (Boyd and Svejcar 2004; Jackson et al. 2006) support a compensatory grazing effect on productivity, and two reports (Popolizio et al. 1994; Jackson et al. 2006) found that exclusion resulted in an accumulation of standing biomass that subsequently suppressed productivity. One report (Kluse et al. 2005) concluded that early-season clipping reduced productivity of two grasses. Three of these studies (Stohlgren et al. 1989; Huber et al. 1995; Cole et al. 2004) concluded that resource availability mediated the effect of grazing intensity on herbaceous productivity. Three studies in *Rangelands* found that late-season clipping had no consistent effect on above- and belowground productivity (Clary 1995, 1999; Martin and Chambers 2002).

> “The peer-reviewed literature generally supports the effectiveness of water developments, supplement placement and herding for reducing riparian vegetation utilization, or time spent in riparian areas”

Cattle grazing a small riparian patch within a sagebrush community. (Photo: Chad Boyd)

We can conclude that there is no universal riparian herbaceous production response to the complex elements (time, intensity, season, or duration) of grazing management. Consequently, what a manager learns on one site may not be transferable to another site.

Hypothesis 3: Livestock Distribution Practices Reduce Time Spent in Riparian Zones or Riparian Vegetation Utilization.

The peer-reviewed literature generally supports the effectiveness of water developments, supplement placement and herding for reducing riparian vegetation utilization, or time spent in riparian areas. Bailey (2004, 2005) and George et al. (2007) have reviewed practices that attract livestock to underused areas and away from riparian habitats. Abiotic and biotic characteristics of landscapes and pastures influence the effectiveness of these practices. A few studies document the effectiveness of drinking-water developments, herding, and strategic placement of supplemental feeds for reducing grazing use and the time spent in riparian areas. Nine out of 10 studies (seven peer reviewed, one thesis, and two in rangelands) report that development of off-stream stock water reduces grazing use or time spent in riparian areas. Six of these studies were conducted in eastern Oregon. McGinnis and McIver (2001) reported that the extent to which livestock can be enticed away from riparian areas depends on season, topography, vegetation, weather, and behavioral differences among animals. Ehrhart and Hansen (1998) evaluated ecological function on 71 streams in Montana and found that off-stream water developments resulted in improved ecosystem health. A few studies have shown that most grazing use occurs within 400 m of stock water sources (Pinchak et al. 1991). Thus, water developments placed at this distance or beyond may be more effective at reducing livestock use in riparian areas than closer installations. Two studies in California (McDougald et al. 1989; George et al. 2008) and one in Montana (Bailey et al. 2008a) have demonstrated the effectiveness of strategic supplement placement for attracting livestock away from riparian areas, and one study in Montana documented the effectiveness of herding with or without supplementation for reducing grazing use in the riparian area. Additional studies in Montana have shown the effectiveness of supplement as a cattle attractant. One study in Nevada documented the effectiveness of shade structures for reducing riparian use. The results of these studies are reinforced by studies in California and Montana that found that riparian health was related to time invested in management by the landowner or manager (Erhart and Hansen 1998; Ward 2002; Ward et al. 2003).

Most of the data supporting these findings come from Oregon (Great Basin or forest), California (oak-woodland and annual grassland), or Montana (plains). We conclude from these studies that water developments, strategic supplement placement, and herding can effectively reduce time spent in riparian zones and riparian vegetation use by livestock. Because the effectiveness of these practices is often controlled more by abiotic (topography and distance from water) landscape characteristics than by biotic characteristics, we believe that they can be generalized to other rangeland ecosystems. Livestock attraction practices work best on gentle slopes and become less effective as slope increases. Narrow riparian corridors that are bound by steep slopes with limited available high-quality forage or water are generally not good candidates for these practices.

Hypothesis 4: Under Initially Degraded Conditions, Grazing Exclusion Can Promote Recovery of Riparian Plant Community Composition.

The peer-reviewed literature generally supports the hypothesis that grazing exclusion can promote recovery of riparian plant community composition in degraded riparian systems. Fencing and use exclusion are commonly used to remove grazing from riparian areas permanently or during recovery periods. Many reports of the impacts of grazing on riparian areas and associated aquatic ecosystems come from comparisons of grazed and ungrazed areas (Larsen et al. 1998; Sarr 2002). Working in north-central Colorado on montane riparian areas, Popolizio et al. (1994) showed that long-term grazing altered plant community composition and cover characterized by more bare ground, dandelion (*Taraxacum officinale*), and clover (*Trifolium repens*) compared to ungrazed areas. Similar findings were reported

by Schulz and Leininger (1990) within the riparian zone bordering Sheep Creek in north-central Colorado. Compositional changes from forb- or nonnative grass–dominated communities toward native grass– and sedge-dominated communities have been widely documented in montane riparian meadow with grazing exclusion (Leege et al. 1981; Kauffman 1983b; Schulz and Leininger 1990; Green and Kauffman 1995).

We can conclude from these studies that grazing exclusion can promote recovery of initially degraded riparian plant community composition. However, plant species richness has not shown a clear response to grazing exclusion, though a few experimental results have been reported in the peer-reviewed literature (Bowns and Bagley 1986; Green and Kauffman 1995).

Hypothesis 5: Livestock and Other Large Herbivores Modify Structure and Composition of Woody Plant Communities.

The literature clearly indicates that livestock and native ungulates can modify the structure and composition of woody plant communities in riparian habitats. The vast majority of papers dealing with woody plants were from the western and northwestern United States; work from the southwestern United States was limited, and southern Plains publications were lacking. Fourteen of 16 papers (Appendix V) indicated structural or compositional modification of woody plant communities as a result of livestock grazing (Green and Kauffman 1995; Samuelson and Rood 2004; Holland et al. 2005). Papers by Sedgwick and Knopf (1991) and Lucas et al. (2004) did not clearly show structural or compositional effects of grazing on woody plant communities. Two papers indicated negative effects of deer (*Odocoileus hemionus*) browsing (Opperman and Merenlender 2000; Matney et al. 2005), two papers indicated negative impacts of elk (*Cervus canadensis*) or moose (*Alces alces*) herbivory (Kay 1994; Zeigenfuss et al. 2002) on woody plants, and Case and Kauffman (1997) reported reductions in woody plant abundance as a result of combined deer and elk herbivory.

Establishment and maintenance of woody plants can be associated with episodic disturbance events (Rood et al. 2007; Bay and Sher 2008); therefore, evaluation of the effects of grazing on establishment and maintenance of woody plants should ideally occur over a sufficient time interval to encompass critical disturbance events. Auble and Scott (1998) reported that recruitment of cottonwood decreased with cattle grazing but that recruitment was highly dependent on infrequent high flow conditions that created suitable habitat for seedlings. Conversely, Sedgwick and Knopf (1991) thought grazing to be a relatively minor impact on willows (*Salix* spp.) and cottonwoods (*Populus* spp.) in comparison to periodic catastrophic flooding (which washed out woody plant habitat). Manoukian and Marlow (2002) concluded that willow canopy cover fluctuated along streams from 1942 to 1985 but that the trend was upward in a USDA Forest Service grazing allotment. They concluded that extended periods (>3 yr) of rest were not necessary for willow recovery if livestock or wildlife use was closely controlled. In many cases, livestock use of woody plants may constitute only a portion of total use when native ungulates are considered. For example, Kay (1994) reported that tall willows had disappeared from 41 of 44 historical photo sets in Yellowstone National Park in association with elk and moose herbivory. Grazing can also affect woody plants through alterations in site hydrology. Such alterations may take the form of direct alterations in physical characteristics of the stream channel associated with changes from high- to low-root-density vegetation as discussed under hypothesis 8. These modifications could indirectly decrease site availability for riparian woody plants by decreasing available water.

The influence of livestock on woody plant structure is complex and dependent on a variety of management and environmental site factors. Hypothesis 3 makes clear that livestock usage of riparian areas is variable and predicated on a variety of management and environmental factors. From hypothesis 5, we can conclude that livestock and other large herbivores can modify the structure and composition of woody plant communities, but the impacts of livestock on woody plant resources are likely to be highly variable from location to location and within a given location over time.

> “Based on these studies, there is sufficient evidence to conclude that fall livestock use of riparian areas can lead to increased utilization of woody plants”

Hypothesis 6: Late-Growing-Season Livestock Use Increases Utilization of Woody Plants. Eleven of the 17 papers associated with livestock impacts on woody plants reported on the effects of late-season use. Of those 11 papers, nine reported negative structural or compositional modification associated with late-season livestock utilization of woody plants (Schulz and Leininger 1990; Clary et al. 1996; Holland et al. 2005), and four papers specifically noted increased use during the late-season period (Roath and Krueger 1982; Kauffman et al. 1983a; Conroy and Svejcar 1991; Green and Kauffman 1995). One paper found that dormant-season clipping had less negative impact on woody plant abundance than continuous elk use (Zeigenfuss 2002), and another paper reported decreased willow abundance associated with late-season deer use (Matney et al. 2005). Roath and Krueger (1982) reported an inverse relationship between degree of woody plant utilization and phenological maturity of herbaceous cover. Kauffman et al. (1983a) and Matney et al. (2005) noted that woody plant utilization by mule deer did not begin until herbaceous availability became limiting. Clary et al. (1996) concluded that spring grazing was less detrimental to woody plants than fall grazing.

Based on these studies, there is sufficient evidence to conclude that fall livestock use of riparian areas can lead to increased utilization of woody plants. This temporal pattern of woody plant utilization is generally associated with reduced herbaceous plant availability or forage quality.

For streams with the site potential to support riparian woody plants, consistent late season grazing can reduce woody plant recruitment and extent. (Photo: Ken Tate)

Hypothesis 7: Riparian Burning Can Reduce Undesirable Woody Species and Restore Desired Herbaceous or Woody Vegetation. Few studies have addressed the effects of fire on riparian ecosystems (Dwire and Kauffman 2003). However, riparian species exhibit adaptations that facilitate rapid recovery following fire. Several species resprout following fire, including quaking aspen (*Populus tremuloides*), cottonwood, and willows. To the extent that fire can remove competition from undesirable species, desirable resprouting species may be restored.

Reviews by Dwire and Kaufman (2003) and Pettit and Naiman (2007) point out that the effectiveness of riparian burning may be mediated by resource availability and grazing management. Riparian burning is not well studied, but these reviews offer several hypotheses related to interactions among climate, disturbance regime, landscape position, and fire frequency and intensity. They point out that even in fire-driven landscapes (e.g., savannas), riparian plant community composition and productivity is more likely to be controlled by water and nutrient availability afforded by the lower landscape position. The effects of burning will likely interact with grazing management with higher grazing intensities, reducing the effects and the likelihood of fire in riparian zones (Dwire et al. 2006). That said, if sufficient fuel is available, the effects of burning may depend on depth to the water table. Blank et al. (2003) burned riparian sites dominated by big sagebrush (*Artemisia tridentata*) to reduce its cover and favor herbaceous species in areas with shallow and deep water tables. Herbs that were present at the time of burning resprouted and contained higher nutrient concentrations following burning. However, herbs were less abundant where the water table was deeper prior to burning, so the postburn response was more favorable with shallower water tables. Proportionally more of the surface soil nutrients were lost from the riparian zones with deeper water tables, which does not bode well for the growth of herbs in these habitats. Thus, recovery from fire depends on the presence of residual herbaceous species and an adequate water table to support these species.

TABLE 3. Grazing system compatibility with willow-dominated plant communities in riparian habitats (Kovalchik and Elmore 1991).

Grazing practice	Compatibility with willows
Corridor fencing	Highly
Riparian pasture	Highly
Spring (early-season) grazing	Highly
Winter grazing	Highly
Two-pasture rotation	Moderately
Three-pasture rest rotation	Moderately
Three-pasture deferred rotation	Moderately
Spring–fall pastures	Incompatible
Deferred grazing	Incompatible
Late-season grazing	Incompatible
Season-long grazing	Incompatible

Literature concerning use of prescribed fire to control undesirable woody species in riparian zones relates mainly to the genus *Tamarix*. Three of four studies that incorporated fire as a treatment reported successful control of *Tamarix* with mortality rates up to 95% (McDaniel and Taylor 2003; Harms and Hiebert 2006; Bateman et al. 2008; Appendix VII). One study found that control of *Tamarix* was not related to burning or mechanical removal but instead was most closely associated with site and year factors, the most important of which was precipitation, with no *Tamarix* regrowth occurring on sites receiving less than 20.8 cm of annual precipitation (Bay and Sher 2008). Busch and Smith (1993) urged caution in the use of fire to control both *Tamarix* and Tessaria, as these genera possess ecophysiological adaptations that may favor their abundance over historically dominant willow and cottonwood in the postfire environment. A study in the Great Basin on riparian areas affected by stream incision and decreased water tables examined the use of prescribed fire to remove sagebrush and restore riparian obligate herbaceous species. Desired plant species increased and herbaceous biomass tripled on sites with residual riparian species and adequate water tables, but sites that

lacked residual species and had significantly lowered water tables were dominated by annual weeds postfire (Chambers and Linnerooth 2001; Wright and Chambers 2002; Blank et al. 2003). Fire has been used as an effective form of control for nonsprouting conifer species on upland sites (e.g., Bryant et al. 1983; Engle and Stritzke 1995; Miller et al. 2005) and may play an important role in controlling encroachment of those species into riparian areas.

While there are insufficient experimental data to thoroughly evaluate this hypothesis, current literature suggests that fire can be used to control some species of woody plants; however, the success of fire-based restoration may relate strongly to the availability of propagules of desired species, which can be depleted if riparian degradation has led to decreased soil water availability.

Scientific Uncertainty and Livestock Exclosure Studies

Comparison of grazed areas with ungrazed areas (exclosures) is a common practice that has the potential for erroneous interpretations. Sarr (2002) reviewed exclosure studies and reported that exclosure-based research has left considerable scientific uncertainty because of the popularization of relatively few studies, weak study designs, a poor understanding of the scales and mechanisms of ecosystem recovery, and selective, agenda-laden literature reviews advocating for or against public lands livestock grazing. Exclosures are often too small (<50 ha) and improperly placed to accurately measure the responses of aquatic organisms or geomorphic processes to livestock removal. Depending on the site conditions when and where livestock exclosures are established, postexclusion dynamics may vary considerably. Systems can recover quickly and predictably with livestock removal, fail to recover because of changes in system structure or function, or recover slowly and remain more sensitive to livestock impacts than they were before grazing was initiated. Sarr presents suggestions for strengthening the scientific basis for livestock exclosure research, including 1) incorporation of meta-analyses and critical reviews, 2) use of restoration ecology as a unifying conceptual framework, 3) development of long-term research programs, 4) improved exclosure placement and design, and 5) a stronger commitment to collection of pretreatment data. Properly designed exclosure studies could provide useful insights into grazing effects, but few meet these criteria.

Practices That Protect or Restore Stream Bank and Riparian Soil Stability

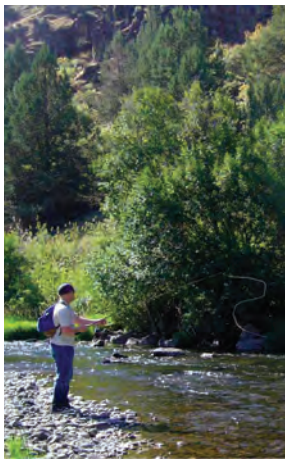
Hypothesis 8: Riparian Management That Affects Plant Species Composition, Plant Vigor, Rooting Densities and Depth, and Ground Cover and Influences the Stability of Stream Channel and Riparian Soils That Derive Their Stability from Riparian Vegetation. The linkage between riparian management, riparian vegetation, and stream channel and riparian soil stability is complex (Fig. 1). Stream systems themselves are complex; Rosgen (1994) describes almost 100 stream channel categories. State factors such as watershed size, geomorphology, parent material, climate, and site-specific riparian vegetation attributes interact to define the structure and function of each stream segment (reach). Each stream reach may support and be supported by different riparian plant communities. Each reach may respond differently to watershed- or landscape-scale disturbances, and each may exhibit differing response to riparian management practices. Gordon et al. (1992), Leopold (1994), and Rosgen (1996) are excellent applications of our basic understanding of stream hydrology and applied river morphology from a watershed perspective.

Eight studies and six reviews provide evidence of the importance of riparian plant communities and grazing management to stream bank and soil stability. Thorne (1982, 1990), Gregory (1992), and Trimble and Mendel (1995) discuss and document the general importance of riparian plant communities on stream channel and riparian soil stability. In the Sierra Nevada, Michelli and Kirchner (2002a) found that the 50-yr rate of stream channel migration and erosion was 6 and 10 times lower on stream banks and associated "wet" riparian areas covered with sedge (*Carex* spp.) and rush (*Juncus* spp.) compared to grass-dominated "dry" stream banks and associated riparian areas. In a companion study, Michelli and Kirchner (2002b) found that the tensile strength of wet

riparian soils supporting sedge (*Carex* spp.) and rush (*Juncus* spp.) plant communities were five times stronger than dry riparian soils dominated by grass and shrub species. Soil tensile strength was positively correlated to plant density, biomass, and the ratio of root to soil mass. In northwestern Nevada, Manning et al. (1989) compared root mass and root length density across a soil moisture gradient represented by four herbaceous riparian plant communities. They found both root metrics to increase with soil moisture availability, indicating superior site-stabilizing capacity in the wetter plant communities. Klienfelder et al. (1992) report similar findings for riparian areas from central Nevada and eastern California.

Research introduced to test hypotheses 1, 2, 4, and 5 in this document establishes the capacity for riparian grazing, along a gradient of heavy to minimal grazing, to have primary effects on riparian plant composition, biomass, and cover. Combined with the discussion above, the capacity for riparian grazing management practices to have secondary effects on stream channel and riparian soil stability can be established. In general, it has been well documented (e.g., Kauffman and Krueger 1984; Sarr 2002) that incorrectly managed livestock grazing in riparian areas can 1) reduce plant root mass and rooting depth, which is critical for stabilizing riparian soils and stream banks against stream flow, and 2) shift plant community composition from high-root-density species to low-root-density species. The subsequent potential effects include: 1) stream banks and riparian soils becoming unstable; 2) stream channels with "hard" bottoms eroding laterally and widening; and 3) stream channels with "soft" bottoms eroding vertically to down-cut the channel and lowering the riparian water table. A positive feedback loop exists between lowered water tables and stream bank stability as it becomes increasingly difficult for high-root-density species that require wet habitats to reestablish (Toledo and Kauffman 2001). There are significant limitations with the literature addressing grazing impacts on stream channel and riparian soil stability and associated ecosystem services (e.g., aquatic habitat, flood attenuation). Several comprehensive reviews of essentially the same literature base substantiate the generally negative effects of "heavy" grazing, the generally positive response of riparian areas to complete removal of heavy grazing, the need for further research on "proper" grazing management strategies for riparian areas, and the need for increased rigor and consistency in case studies and experiments examining these riparian grazing strategies (Rinne 1988; Platts 1991; Ohmart 1996; Larsen et al. 1998; Allen-Diaz et al. 1999; Sarr 2002).

Based on these reviews and studies, there is sufficient evidence that riparian grazing management that maintains or enhances key riparian vegetation attributes (i.e., species composition, root mass and root density, cover, and biomass) will enhance stream channel and riparian soil stability, which will in turn support ecosystem services, such as flood and pollutant attenuation and high-quality riparian habitat. Lacking in the literature are watershed-level, statistically robust examinations of how stream channel and riparian soil stability are correlated with grazing management components, such as intensity, frequency, season, and duration of grazing across a set of riparian conditions. These should be compared for a variety to conditions, including degraded and undegraded riparian systems, herbaceous-dominated and woody-dominated systems, and alluvial channel substrates versus bedrock-dominated substrates. It is difficult to predict the specific impacts of riparian grazing management practices under differing levels of state variables (Fig. 1) as indicated in the results of Lucas et al. (2004) and Jackson and Allen-Diaz (2006) for hypothesis 1. However, it is clear that riparian grazing can be managed to enhance and protect primary riparian vegetation attributes that are strongly correlated to stream channel and riparian soil stability, which support ecosystem services provided by riparian areas (e.g., hypotheses 3 and 4). There may be highly degraded riparian conditions, such as down-cut channels, under which riparian grazing management practices alone cannot restore the site's former soil moisture regime that supported riparian plant communities with high rooting densities and their associated ecosystem services (Chambers and Miller 2004).



Riparian corridors are important for recreation. (Photo: Chad Boyd)

Healthy riparian plant communities are effective at attenuating pollutants carried in runoff. (Photo: Ken Tate)

Practices That Protect or Enhance Ecosystem Services

Hypothesis 9: Riparian Vegetation Can Attenuate Pollutants Transported in Runoff, and Buffer Strip Effectiveness Is Dependent on Site-Specific Factors. The management of riparian vegetation to trap waterborne pollutants is commonly referred to as a vegetative "buffer" or "filter" strip. Small wetlands, either natural or constructed, can also provide this service. Reviews of research relevant to the implementation of vegetative buffers in riparian habitats of rangeland ecosystems and pastures can be found in Castelle et al. (1994), Schmitt et al. (1999), Dosskey (2002), Dorioz et al. (2006), and Mayer et al. (2007), among others.

Attenuation efficiencies ranging from ~0 to greater than 99% have been reported for pollutants common to rangelands and livestock-grazed systems, primarily nutrients, sediment, and indicator bacteria and pathogens (Dillaha et al. 1989; Pearce et al. 1998b; Atwill et al. 2002, 2005; Bedard Haughn et al. 2004; Tate et al. 2004a, 2005; Dosskey et al. 2007; Knox et al. 2007, 2008). The variation observed across these studies can partially be attributed to site-specific differences in biophysical factors, such as buffer width, slope, vegetation attributes within the buffer, pollutant type and attributes, pollutant load entering the buffer, overland or flood flow rate entering the buffer, hydrologic residence time within the buffer, riparian soil attributes within the buffer, and

buffer vegetation management (Castelle et al. 1994; Schmitt et al. 1999; Mayer et al. 2007). A significant number of studies have focused on determination of optimal buffer widths. No single buffer width can be prescribed for all scenarios, and there is increasing demand for decision support tools that develop first approximations of required buffer widths based on site factors (e.g., Dosskey et al. 2005, 2006, 2008; Parajuli et al. 2008). While biophysical site factors determine buffer efficiency, the manager must also decide on an acceptable level of water quality degradation risk in the determination of buffer width. As risk tolerance decreases, buffer width must increase (Castelle et al. 1994; Atwill et al. 2005; Tate et al. 2005). Varying results have been reported for the effect of stubble height of herbaceous riparian vegetation on sediment and nutrient deposition and retention, indicating that this metric may not consistently impact, or predict, buffer efficiency (Abt et al. 1994; Clary et al. 1996; Pearce et al. 1997, 1998a, 1998b; Thorton et al. 1997; Fraiser et al. 1998; Skinner 1998; Clary and Leininger 2000; Marlow et al. 2006). There is a consistently positive correlation between vegetative ground cover, plant stem density, and buffer filtration efficiency for several pollutants (e.g., Larsen et al. 1993; Corley et al. 1999; McEldowney et al. 2002; Davies et al. 2004; Tate et al. 2005). It is important to note that these same plant attributes are important for determining stream channel and riparian soil stability (H 8). Defoliation to manage buffer vegetation biomass accumulation, growth stage, and nutrient demand affects the nitrogen attenuation efficiencies of buffers (e.g., Mendez et al. 1999; Matheson et al. 2002; Bedard-Haughn et al. 2005; Jackson et al. 2006). As overland and flood flow rates entering a buffer increase and hydrologic residence times decrease, buffer attenuation and retention capacities can be reduced, if not completely eliminated (e.g., Bedard-Haughn et al. 2004; Tate et al. 2004a, 2005; Knox et al. 2007, 2008). Biomass accumulation in buffers can create human health concerns. Excessive organic carbon near surface drinking water sources may lead to formation of carcinogenic–mutagenic by-products during chlorination (Krasner et al. 1989; Jassby and Cloern 2000; Bull 2001).

Based on 41 peer-reviewed reports, the overriding message is that 1) vegetative buffer strips can attenuate some portion of most waterborne pollutants transported by overland and flood flow events, and 2) there is significant variation in buffer attenuation efficiency attributable to site-specific factors. Supporting research ranges across a wide range of systems (e.g., urban, agricultural, rangeland), regions of the United States and the world and for a wide suite of pollutants, including sediment, nutrients, microorganisms, and pesticides. There is strong evidence supporting the overall assertion that riparian vegetation can function to attenuate waterborne pollutants in overland and flood flow events.

Hypothesis 10: Practices That Reduce Livestock Densities, Residence Time, and Fecal and Urine Deposition in Riparian Areas and Stream Flow Generation Areas Can Reduce Nutrient and Pathogen Loading of Surface Water. In conjunction with implementation and management of vegetative buffers in riparian areas, additional water quality protection can logically be derived from implementation of livestock management strategies that distribute livestock fecal material and urine away from riparian areas, stream flow generation areas, and surface waters. In essence, this will create additional buffering length and capacity. Recent research on rangelands supports that livestock distribution practices can be applied to modify the spatial distribution of feces and urine deposition, creating buffering distances between feces and water bodies with minimal establishment of fences (Miner et al. 1992; Clawson 1993; Bailey et al. 1996; Bailey and Welling 1999; Bailey et al. 2001; Tate et al. 2003; Blank et al. 2006; Bailey et al. 2008a, 2008b). Cattle feces and urine distribution patterns on rangelands are significantly associated with location of livestock attractants, aspect, topographic position, and season (Tate et al. 2003; Bailey et al. 2008a). Strategic location of livestock attractants, including stock water, mineral supplements, and protein supplements, can have strong influences on patterns of cattle fecal and urine loads on watersheds.

There is evidence to support the assertion that practices that reduce livestock densities, residence time, and fecal and urine deposition in riparian areas and stream flow generation

> “There is strong evidence supporting the overall assertion that riparian vegetation can function to attenuate waterborne pollutants in overland and flood flow events”

> We conclude that grazing can decrease populations of riparian obligate avifauna but has variable effects on generalist species"

areas can reduce nutrient and pathogen loading of surface water. In addition, reducing livestock densities and residence time also reduces the negative effects of livestock on riparian vegetation attributes (e.g., stem density, cover) and soil hydrologic attributes (Gifford and Hawkins 1978; Tate et al. 2004b; e.g., bulk density, infiltration capacity). Thus, reducing livestock densities and residence time can benefit buffer efficiency (H 9) and site stability (H 8). To reduce livestock impacts in the riparian zone, stocking rate reductions are not the universal solution. The key is implementing practices that reduce livestock density in the riparian zone. These can be distribution practices, fencing to increase control of time and duration of grazing, and, in some cases, stocking rate reductions.

Hypothesis 11: Riparian Grazing Decreases Habitat Quality for Prairie Wetland Avian Species. Nine of 28 avian references provided information on grazing management of wetland or prairie wetland habitat and associated avian species. In seven studies focusing on waterfowl (Duebbert et al. 1986; Ignatiuk and Duncan 2001; Murphy et al. 2004), only four had ungrazed controls. Habitat quality was unchanged in two studies (Barker et al. 1990; West and Messmer 2006), decreased in a third (Kruse and Bowen 1996), and was unreported in a fourth (Littlefield and Paulin 1990); nesting success was unchanged, decreased, increased, or was not measured. One of three studies without controls indicated that heavy stocking rates did not provide adequate nesting cover (Duebbert et al. 1986), and two studies found no difference in nest success or habitat quality between season-long and rotational grazing strategies (Ignatiuk and Duncan 2001; Murphy et al. 2004). One of two passerine studies indicated decreased habitat quality and bird diversity with grazing (Taylor 1986), and a second study found that avian abundance and diversity were unaffected by grazing (May et al. 2002).

The references reviewed here suggest that with the exception of heavily grazed areas, grazing in wetland habitat does not decrease habitat quality for waterfowl. Insufficient data exist to determine the influence of grazing on habitat quality for wetland passerine species.

Hypothesis 12: Riparian Grazing Decreases Populations of Riparian Avifauna. Effects of livestock grazing on riparian avian habitat have been reviewed or summarized (Szaro 1980; Bock et al. 1993; Fleischner 1994; Belsky et al. 1999). The importance of riparian vegetation as avian habitat has been described by numerous authors (Bull and Skovlin 1982; Douglas et al. 1992; Sanders and Edge 1998; Deschenes et al. 2003). Knopf et al. (1988a) reported that riparian vegetation attracts over 10 times the number of spring migrant birds found in upland sites and has 14 times more species during fall migration. References were fairly well distributed geographically except literature for the southern Plains, which was generally lacking. Evaluating the influence of grazing management practices on riparian wildlife was limited by insufficient details in many of the studies reviewed. These limitations relegated our assessment of grazing responses to a presence-and-absence standpoint.

Nineteen studies report dynamics of riparian avifauna as a function of grazing. Of those, eight found no change in abundance (Kauffman et al. 1982; Sedgewick and Knopf 1987; Knopf et al. 1988b; Schulz and Leininger 1991; Warkentin and Reed 1999; Stanley and Knopf 2002; Scott et al. 2003; Martin and McIntyre 2007), five did not report or did not clearly report abundance (Neel 1980; Crawford et al. 2004; Martin et al. 2006; Brodhead et al. 2007; Hall et al. 2007), and five found decreased abundance (Popotnik and Giuliano 2000; Tewksbury et al. 2002; Krueper et al. 2003; Earnst et al. 2005; Fletcher and Hutto 2008). Four studies reported a decrease in species diversity or richness of riparian avifauna (Popotnik and Giulano 2000; Stanley and Knopf 2002; Scott et al. 2003; Hall et al. 2007), and four reported static values (Kauffman et al. 1982; Schulz and Leininger 1991; Warkentin and Reed 1999; Earnst et al. 2005). Bock et al. (1993) reviewed abundance data for 63 neotropical migrant bird species in grazed and ungrazed environments. Of these species, three declined in abundance in grazed areas, and seven additional species were thought to be negatively influenced by grazing. These species were either shrub, ground, or near-ground nesters.

Overall, where grazing induced changes in habitat structure and composition, avian populations tended to change from dominance by riparian obligate species to dominance by riparian generalists (e.g., Schulz and Leininger 1991; Martin and McIntyre 2007). Changes in avian abundance were often positively associated with habitat quality; however, assigning a habitat quality measure is somewhat subjective when dealing with avian species assemblages; some species may benefit from altered habitat, and some may be negatively impacted, depending on specific habitat requirements (Farley et al. 1994). The work of Martin and McIntyre (2007) suggests that species diversity may be maximized with heterogeneous grazing intensities over space. Tewksbury et al. (2002) suggested that avian species nesting below 2.5 m would be most negatively impacted by livestock grazing.

We conclude that grazing can decrease populations of riparian obligate avifauna but has variable effects on generalist species. Diversity of species may decrease in proportion to grazing-induced decreases in habitat diversity (Scott et al. 2003). One caveat to this conclusion is that determining the specific influence of grazing on riparian avian assemblages is challenging and must take into account uses and changes in use within the surrounding landscape. Avian species are highly mobile, and some "riparian" species may depend on spatially distant habitat types and landscape attributes. In an extreme example, assessing the influence of management practices on abundance of riparian neotropical migrant avifauna should involve determination of vital rates (e.g., nesting success and juvenile survival) to help factor out the proportion of population change associated with nonbreeding habitat. Management of local-scale riparian issues (such as grazing) should be undertaken in conjunction with larger-scale efforts to create landscapes suitable for attaining conservation objectives for riparian avifauna (Martin et al. 2006; Fletcher and Hutto 2008).

Hypothesis 13: Riparian Grazing Decreases Populations of Macroinvertebrates, Herpetofauna, and Salmonids. Limited data suggest that grazing does not decrease the abundance or overall diversity of macroinvertebrates. However, some habitat specialists may decrease and be replaced with

Grazing should be managed to allow a site to meet its potential to provide in-stream aquatic habitat features such as over-hanging banks and clean gravel beds. (Photo: Ken Tate)

habitat generalists (Weigel et al. 2000; Bates et al. 2007). Data are insufficient to make general conclusions regarding the influence grazing on herpetofauna populations. Diversity of macroinvertebrates remained unchanged or increased with grazing in six of nine studies (Fritz et al. 1999; Weigel et al. 2000; Homyack and Giuliano 2002; Scrimgeour and Kendall 2003; Sada et al. 2005; Bates et al. 2007), decreased in one study (Foote and Rice Hornung 2005), and was not reported in two studies (Tait et al. 1994; Saunders and Fausch 2007). Macroinvertebrate abundance remained unchanged with grazing in seven studies (Tait et al. 1994; Fritz et al. 1999; Homyack and Giuliano 2002; Scrimgeour and Kendall 2003; Sada et al. 2005; Bates et al. 2007; Saunders and Fausch 2007), decreased in one study (Foote and Rice Hornung 2005), and was not reported in one study (Weigel et al. 2000). Two studies reported no effect of grazing on riparian herpetofauna (Bull and Hayes 2000; Homyack and Giuliano 2002), but a review by Brodie (2001) suggests that turtle populations may be negatively impacted by increased siltation associated with disturbances, such as livestock grazing.

Limited data suggest that livestock grazing can decrease salmonid populations, and the bulk of papers we examined suggested decreasing quality of habitat with livestock use. The specific grazing management scenarios under which salmonid populations may be negatively impacted by grazing are largely unknown given that most of the salmonid studies we reviewed did not report stocking rate or utilization information. Impacts of livestock grazing on salmonid habitat and populations have been summarized (Meehan and Platts 1978; Platts 1981, 1991; Armour et al. 1994; Fleischner 1994; Belsky et al. 1999). Three of six studies reported decreased salmonid abundance associated with livestock grazing (Keller and Burnham 1982; Tait et al. 1994; Knapp and Matthews 1996), one study reported no impact (Chapman and Knudsen 1980), and one did not report abundance as a function of grazing treatment (Platts and Nelson 1989). One study indicated that salmonid abundance was higher for areas grazed with a high-density, short-duration grazing system compared to season-long grazing (Saunders and Fausch 2007). Three of six studies reported decreased quality of salmonid habitat with grazing (Chapman and Knudsen 1980; Platts and Nelson 1989; Knapp and Matthews 1996), one reported no effect (Tait et al. 1994), one did not report habitat effects (Keller and Burnham 1982), and one reported increased habitat quality with short-duration grazing compared to season-long grazing (Saunders and Fausch 2007).

We recognize that additional published references are available correlating fish abundance with grazing practices. However, much of this work has not undergone the scrutiny of peer review, suffers from major experimental design inadequacies (e.g., lack of replication, nonrandom treatment assignment, lack of pretreatment data), or has insufficient methodological description to determine the adequacy of experimental design (Platts 1982; Rinne 1985; Larsen et al. 1998). These problems render affected references useless for our purposes in determining the validity of hypotheses regarding management practices. That said, it should also be pointed out that ill-advised grazing practices can lead to loss of bank-stabilizing vegetation, resulting in altered channel morphology (see discussion for hypothesis 8) and that such alterations may have strong negative consequences for habitat of affected aquatic fauna (Fitch and Adams 1998).

Hypothesis 14: Riparian Grazing Decreases Habitat Quality for Riparian Mammals.

Data are insufficient to determine the impact of grazing on large mammal riparian wildlife species with two studies reporting either decreased quality of fawning habitat (Loft et al. 1987) or livestock-induced habitat avoidance (Loft et al. 1991). Three studies addressed the influence of grazing on riparian small mammal communities. Two studies found no change in diversity of species (Kauffman et al. 1982; Schulz and Leininger 1991), and one reported decreased diversity (Giuliano and Homyak 2004). Two of three studies reported decreased small mammal abundance with grazing (Kauffman et al. 1982; Giuliano and Homyak 2004), and one study was inconclusive (Schulz and Leininger 1991). A fourth study reported decreased small mammal biomass with heavy compared to light grazing, but responses varied by species, and ungrazed comparisons were not included (Johnston and Anthony 2008).

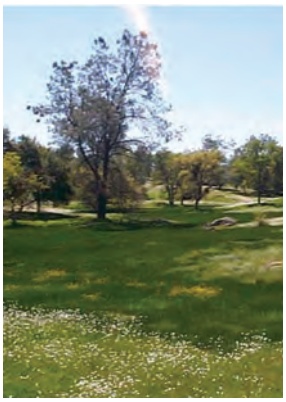
Available data are insufficient to draw general conclusions regarding the impacts of livestock on riparian small mammal communities. Realistically, a general conclusion for small mammals as a group may not be possible because of the inherent variability in habitat requirements between species and variability among years. Species that depend on herbaceous cover may decrease with heavy livestock use, while the same disturbance may increase habitat quality for species requiring reduced amounts of herbaceous ground cover (Hanley and Page 1982; Johnston and Anthony 2008).

Hypothesis 15: Grazing Removal Will Increase Quality of Sage Grouse Brood-Rearing Habitat. Because of a lack of experimental work on the subject, generalizations regarding the influence of grazing on sage grouse brood rearing habitat cannot be made at this time. Only one study, Neel (1980; see also Appendix VI) has addressed the influence of grazing on riparian brood-rearing habitat for sage grouse. This study found that 1 yr of rest from grazing increased abundance of forbs important in the diet of sage grouse. The author also reported that sage grouse selected lightly grazed riparian habitat for brood rearing as compared to nongrazed habitat.

Hypothesis 16: Invasive Woody Plant Management Can Control Abundance of Undesirable Plant Species. Literature relating to invasive species management and riparian woody plants deals mainly with populations of the invasive plant genera *Tamarix* and in some cases Russian olive (*Elaeagnus angustifolia*) in the southwestern United States. In three studies, various combinations of cutting, plowing, and burning were highly effective at removing both *Tamarix* and Russian olive (McDaniel and Taylor 2003; Harms and Hieber 2006; Bay and Sher 2008; Appendix VII). A fourth study reported that while *Tamarix* seedling density was initially higher than that of native woody plants, *Tamarix* seedlings were much more susceptible to mortality associated with overbank flooding in unregulated river systems (Sher et al. 2002). Much of the effort to control *Tamarix* remains unevaluated and unpublished. Bay and Sher (2008) reviewed control projects ranging from 1 to 18 yr posttreatment. They reported that the degree of control was not related to time since restoration began or specific management treatments and that areas with less than 21 cm of annual precipitation had only limited long-term *Tamarix* control. Site factors played a strong role in influencing project success, and the degree of control was associated positively with proximity to perennial water, sufficient precipitation, recent flooding, and coarse soil texture. Shafroth et al. (2008) noted that success of *Tamarix* control projects was highly variable and proposed a framework for planning control efforts that focuses on using principles of adaptive management. These authors stressed that site conditions, including soil salinity and texture, current vegetation, and availability of desired propagules, have a strong influence on restoration success and highlighted the importance of considering both passive (e.g., flooding) and active (e.g., cutting and seeding) management options.

Because of the spatial and temporal variability associated with the success of *Tamarix* control projects, it is not possible to make general statements regarding the effectiveness of control programs. Future success in *Tamarix* management will likely hinge on effective application of adaptive management techniques (Reever Morghan et al. 2006).

Hypothesis 17: Control of Invasive Woody Plant Species Increases the Abundance of Terrestrial Wildlife. Most of the literature regarding invasive riparian woody plant control and wildlife abundance relates to the control of *Tamarix*. Four papers relating invasive woody plants to wildlife assemblages failed to uncover substantive benefit to abundance or diversity of avian, butterfly, or lizard assemblages (Knopf and Olson 1984; Bateman et al. 2008; Nelson and Wydoski 2008; Sogge et al. 2008). In a 2008 review, Sogge et al. found that not all avian species benefit from control of *Tamarix*, particularly when native vegetation does not reestablish in the postrestoration environment. These authors concluded that 49 avian species, including the endangered southwestern willow flycatcher, use *Tamarix* as breeding habitat. Van Ripper et al. (2008) reported that for most avian species, abundance was highest with a mix of native woody plants and *Tamarix*.



Seasonal riparian corridor in an oak-woodland in California's central Sierra Nevada foothills. (Photo: Melvin George)

Knopf and Olson (1984) reported increased avian diversity in native riparian communities compared to sites dominated by Russian olive but noted that Russian olive was used by avian species favoring tall shrub habitat and that the occurrence of Russian olive near the periphery of riparian areas could increase diversity of riparian habitats available to avian species.

At this time, the complexity of wildlife responses to *Tamarix* and Russian olive control varies strongly across species and geographic location, making generalizations regarding the impact of these invasive species on terrestrial wildlife difficult (Shafroth et al. 2005; Sogge et al. 2008).

Hypothesis 18: Upland Brush Management Can Decrease Riparian Erosion and Increase Stream Flow. Nine studies addressed the influence of woody plant removal on watershed hydrology (Appendix VII). Three studies reported increased water yield (actual or modeled) or stream flow in pinyon-juniper (Baker 1984), chaparral (Davis 1993), or sagebrush (Sturges 1994) vegetation, and two reported no change in stream flow (Wilcox et al. 2005) or runoff (Dugas et al. 1998) for Ashe juniper (*Juniperus ashei*). One study found no change in basin-level water yield with removal of western juniper (*Juniperus occidentalis*; Kuhn et al. 2007). One study reported increased potential for deep drainage with burning in sagebrush-steppe (Seyfried and Wilcox 2006), a second found dramatically decreased runoff and erosion in sagebrush-steppe following juniper removal and (Pierson et al. 2007), and a third found increased runoff with chaining and windrowing of pinyon-juniper, but runoff was invariant when trees were left in place (Gifford 1975). Wilcox (2002) proposed that the influence of woody plants on stream flow will be a product of interactions between shrub characteristics, precipitation, soils, and geology. Under this conceptual framework, woody plant removal generally will not affect stream flow in areas receiving less than 500 mm of annual precipitation, and runoff will occur as overland flow in the absence of a subsurface connection between stream and hillslope. Without subsurface flow, water use by woody plants may have little impact on stream flow. Huxman et al. (2005) echoed the importance of subsurface flow for linking woody plants and stream flow. Pierson et al. (2007) stressed that increases in herbaceous production associated with woody plant control can dramatically increase infiltration and decrease runoff.

Given that site characteristics strongly influence the relationship between woody plant cover and hydrology, definitive statements regarding outcomes of this interaction are not possible. Newman et al. (2006) argued that because of the variability and complexity of the relationship between woody plants and rangeland hydrology, efforts to manage woody plant issues will benefit from "place-based science" and an interdisciplinary focus on hypothesis testing.

Hypothesis 19: Shading of the Stream Channel by Riparian Woody Vegetation Cover Influences Aquatic Ecology by Reducing Stream Temperature. Macroinvertebrates and fish are sensitive to dissolved oxygen content of streams, which is influenced by stream temperature. Thus, stream temperature is an important factor affecting the distribution of aquatic vertebrate and invertebrate species (Baltz et al. 1987; Lyons 1996; Hawkins et al. 1997; Jacobsen et al. 1997; Isaak and Hubert 2001). The distribution and abundance of native coldwater fisheries in the western United States has been reduced since European settlement (Nehlsen et al. 1991; Hunnington et al. 1996; Thurow et al. 1997), and land and water management practices that impact stream temperature are considered to be partly responsible for these reductions (Isaak and Hubert 2001; Poole and Berman 2001; Zoellick 2004). Water temperature is a particularly important habitat determinant for aquatic species in arid rangeland basins of the western United States.

A significant international research base indicates that water temperature is a spatially and temporally dynamic stream property controlled by a complex and interacting set of environmental factors, such as local air mass characteristics, solar radiation, vegetative and topographic shading, channel elevation and aspect, channel gradient, adiabatic rate, channel width and depth, hydrologic residence time, stream flow volume, and deep and shallow groundwater inputs (e.g.,

Constantz 1998; Ebersole et al. 2001; Liquori and Jackson 2001; Poole and Berman 2001; Ebersole et al. 2003; Johnson 2004; Malcolm et al. 2004; Moore et al. 2005; Tate et al. 2007). Management practices which affect these factors have the potential to secondarily affect stream temperature dynamics. There is clear evidence that shading provided by woody plant cover will have some effect on stream temperature dynamics.

There is also clear evidence that the relative importance of woody plant canopy cover, among the many factors, in determining stream temperature is variable across ecosystems, watersheds, streams, and stream reaches. Larson and Larson (1996) hypothesized that when air temperature is warmer than water temperature, water temperature will increase to approach thermal equilibrium with the surrounding air mass and that this basic relationship is unchanged by the presence of shade from woody plants. However, in a field study in arid northeastern California, Tate et al. (2005) found that daily maximum stream temperature was associated with air temperature, instantaneous stream flow volume, stream order and watershed position, and woody plant canopy cover. Increased woody plant cover was associated with decreased maximum daily stream temperature, and a significant interaction between canopy and air temperature indicated that the cooling effect of woody plant cover increased with increased maximum daily air temperature. One study (Meays et al. 2005) reported that a thermal gradient associated with variable elevation was the dominant factor controlling stream temperature and that exposure time (velocity and distance), discharge volume, rate of flow, and cool-water inputs had a greater influence on stream temperature than woody canopy cover. Poole and Berman (2001) found that the influence of shade on stream temperature was greatest in smaller (first and second order) streams and decreased with stream size. These authors hypothesized that reduced stream shading may lower the quantity of air trapped by vegetation, which can increase convective and advective transfer of heat to the stream surface. Both Liquori and Jackson (2001) and Malcolm et al. (2004) determined that the type of riparian woody plant community affected the relationship between canopy cover and stream temperature. Stream channel protection from incoming radiation is one mechanism by which woody plant cover may influence stream temperature. In certain riparian areas, woody plants may play a role in maintaining channel structure (e.g., width:depth ratio) in the face of destabilizing flood flow events (Winward 2000). To the extent that maintenance of channel structure is related to stream temperature dynamics, woody plants may play an important role in moderating in-stream temperature fluctuations (Liquori and Jackson 2001).

There is strong evidence to support the assertion that riparian management to enhance and sustain riparian woody plants can moderate stream temperatures. The validity of this conclusion is conditional and dependent on 1) site conditions and potential and that management of factors such as stream flow volume may have a greater effect on stream temperature than management of woody plant cover; 2) the natural, or existing, potential of the riparian site to support woody plant communities; and 3) the likelihood that management to increase woody canopy above natural site potential can lead to overall reductions of in-stream primary production, diversity, and richness of aquatic species (Liquori and Jackson 2001; Broadmeadow and Nisbet 2004; Malcolm et al. 2004).

Hypothesis 20: Prescribed Fire Can Increase Richness, Diversity, and Abundance of Native Riparian Plant and Animal Species. Because of the low number of published reports concerning prescribed fire in riparian habitat, we included wildfire-based publications in this discussion. Of seven papers documenting the effects of fire on native riparian vegetation, four reported little to no effect (Busch and Smith 1993; Gom and Rood 1999; Blank et al. 2003; Smith et al. 2007), and three reported an increase in desired species (Stein et al. 1992; Kay 1993; Rood et al. 2007; Appendix VII). In some cases, fire has been used as a tool to rejuvenate dense stands of mature woody plants, such as cottonwood, when reproduction became limited (Rood et al. 2007).

“There is strong evidence to support the assertion that riparian management to enhance and sustain riparian woody plant cover or canopies can moderate stream temperatures”

Management which supports woody riparian plants and deep-rooted herbaceous vegetation may increase carbon sequestration in riparian soils. (Photo: Mel George)

Existing literature is not sufficient to generalize the effects of fire on riparian plant species richness, diversity, and abundance. The impact of fire on diversity of riparian animal species in native riparian habitat is practically unexplored. Literature documenting the response of riparian birds to fire is lacking (Bock and Block 2005; Smith et al. 2007).

Hypothesis 21: Carbon Storage Is Enhanced by Establishment and Maintenance of Woody Species, Herbaceous Species with High Root Mass, and Dominance of Deep-Rooted Perennials.

Carbon accumulation in soils occurs when C inputs to the ecosystem as NPP exceed C outputs from the ecosystem as microbial respiration of soil organic matter (Post and Kwon 2000). Hence, any management that increases production and/or decreases microbial respiration on an annual basis should promote soil C storage. Root detritus is a significant contributor to recalcitrant soil C pools (Rees et al. 2005), so promotion of belowground NPP is believed to be particularly important for C sequestration.

Nine to 18 yr of grazing exclusion from herb-dominated wet and dry meadows in eastern Oregon resulted in clear increases in belowground standing biomass (Kauffman et al. 2004). While these authors did not detect significant increases in soil organic matter, bulk densities decreased significantly, and one can infer that soil organic C would increase over a longer period of observation. One year of late-season clipping of mesic meadow species in central Nevada resulted in higher rooting activity in the surface 5 cm (Martin and Chambers 2002). These authors cite the overriding influence of water table depth as the reason that larger defoliation effects were

not observed. No significant effects on soil physical or chemical properties were found by Wheeler et al. (2002) as the result of a one-time intense grazing event. Clary and Kinney (2002) found that simulated season-long, heavy grazing significantly reduced root production, while simulated moderate seasonal grazing had no effect relative to unclipped control plots.

We cannot support or reject this hypothesis because few studies have investigated carbon storage in riparian zones. Two studies (grazing exclusion and late-season clipping) support increases in belowground biomass, one study found no grazing effect on soil chemical and physical properties, and one study found that season-long heavy grazing reduced root production in riparian systems.

Incorporating Science into NRCS Conservation Practices and Systems

The evidence supporting conservation practice effectiveness is mixed with some practices being well documented and others poorly supported in peer-reviewed scientific publications. However, this is not the only source of evidence for practice effectiveness. Professional experience is also an important source of knowledge regarding practice effectiveness. Within NRCS and other agencies are conservationists who have learned to apply practices effectively by learning from others and by trial and error, much the way agricultural producers learn to adapt practices in their farming operations. Management based on trial and error is often called adaptive management. Adaptive management allows managers to monitor and evaluate management practices in the field as they go along. The nine steps of planning used by NRCS make up a form of adaptive management that allows conservationists and landowners to identify resource concerns and alternative practices. Following selection and implementation of practices, monitoring and evaluation provide feedback regarding progress toward objectives and practice effectiveness. The knowledge gained during planning, implementation, and evaluation is seldom reported in peer-reviewed journals. Occasionally, it appears in case study reports, but more often it goes unpublished (e.g., Wyman et al. 2006).

Early in the planning process, NRCS conservationists document and analyze resource concerns, including those related to riparian systems and associated watersheds. This is followed by development of alternative practices that may address concerns. Based on this analysis, the landowner selects a mix of practices. For riparian areas, prescribed grazing (528), off-site water (614), fencing (382), and riparian herbaceous cover (390) are common conservation practices that are often applied together because they facilitate control of riparian use while enabling use of the broader landscape by livestock and wildlife.

While the effectiveness of these practices may not have been documented in the ecosystem or site being managed, there is often support for their effectiveness from other riparian ecosystems in the scientific literature. NRCS training programs expedite integration of results from other ecosystems into the planning process. Conservationists and landowners learn what works from these applications, and it becomes part of the individual's experience and the agency's institutional memory in the form of state practice standards and specifications.

RECOMMENDATIONS

With more than 40 management practices (USDA NRCS 2003) available for application to riparian habitats, considerable overlap exists among the purposes and benefits stated in practice standards. Better riparian practice standards could be developed by the following:

- Initiating review teams of NRCS conservationists, biologists, and engineers to complete practice revisions
- Grouping practices that protect or restore vegetation to remove overlapping purposes (e.g., channel bank vegetation, conservation cover, critical area planting, riparian herbaceous cover, stream bank and shoreline protection, and tree and shrub establishment)
- Grouping and revising buffer and filter strip practices into those that apply to rangeland, forestland, or cropland
- Updating practice definitions, purposes and benefits, criteria and other practice standard sections to reflect current knowledge

> “Rigorous monitoring to document the effect of resource management systems can help resource professionals learn more about riparian processes and management interactions”

- Incorporating ecosystem services into revised practice purposes and benefits
- Separating structural practices from vegetation management practices
- Incorporating riparian purposes and benefits into upland practices, such as brush management, prescribed burning, and prescribed grazing

While there are opportunities to combine and clarify practices and there is evidence supporting the effectiveness of many riparian management practices, we can provide little evidence-based support to USDA NRCS for modifications of existing practice specifications (practice application) or initiation of alternative practices, with one exception in the following paragraph. We also recommend addition of a collaborative research and monitoring component to selected practice implementation plans so that the body of evidence supporting conservation practices and systems of practices can be strengthened.

We found sufficient evidence to recommend that NRCS increase the role of herding and supplement placement along with water development and fences for manipulating livestock distribution. These practices have a role where topography does not limit their effectiveness and total exclusion is not required. While it has not been the policy of USDA conservation cost-share programs to fund feed purchase or herding, placement of supplement and herding practices should be included in the overall ranch conservation plan. The USDA might consider allowing these “feeding” practices to be part of the rancher’s share of the cost in the Environmental Quality Incentives Program and other cost-share programs.

The need for more effective selection and application of management practices on a site-specific basis requires much greater attention. Recognizing that one practice or set of practices cannot meet the conservation requirements of biophysically diverse riparian habitats and stream systems, USDA NRCS applies resource management systems that are a flexible mix of practices selected for a specific set of site conditions and landowner management objectives during conservation planning. Rigorous monitoring to document the effect of resource management systems can help resource professionals learn more about riparian processes and management interactions while maintaining feedback information to both land managers and conservation planners. The portfolio of research and case studies supporting the effectiveness of these practices is limited and commonly cannot be extended to other sites. Time and funds limit the ability of research institutions to investigate the seemingly infinite combination of site conditions that exist across US rangeland riparian zones. To accelerate these investigations, we recommend that a partnership of researchers and NRCS conservationists implement two complementary lines of investigation. In the first line of investigation, this team should 1) develop, implement, and maintain rigorous monitoring of selected practices in selected ecosystems and 2) implement monitoring systems that can be analyzed and meet standards of research peer review. The team would manage monitoring data collection and analyze the data at appropriate time intervals. In short, a research study design (monitoring plan) needs to accompany and be funded along with the conservation plan.

The second line of investigation is to develop testable hypotheses based on observations and findings resulting from team monitoring projects. The team can test these hypotheses in more controlled studies. These hypotheses should attempt to elucidate the intervening ecological processes between practice implementation and practice effect. Only then can we begin to understand relationships between grazing, riparian management practices, and riparian ecology at relevant scales and extrapolate results from one location to another. It is crucial that USDA NRCS and other agencies support such a monitoring partnership between researchers and conservationists.

Finally, we support the completion of riparian ecological site descriptions by USDA NRCS. It is important to recognize that not all riparian areas have the same potential or react to management in the same way. Therefore, they should be managed according to their unique characteristics as described in ecological site descriptions. State variables (soils, climate,

geomorphology, topography, vegetation, and wildlife), vegetation dynamics, and practices that have been effective on the site in the past are described in the ecological site description. We recommend completion of riparian ecological site descriptions as a means of documenting and communicating proven site-specific management practices to the NRCS planning process.

KNOWLEDGE GAPS

Our assessment reveals limited controlled experimentation in support of many of our hypotheses, resulting in critical knowledge gaps across all riparian management practices and riparian ecosystems. Linking conservation planning and management to research in a collaborative program is crucial to filling these knowledge gaps.

Two substantial knowledge gaps exist in the riparian literature related to grazing and rangeland management. While there are many case studies comparing species (animal and plant) abundance within and outside riparian exclosures, they are often deficient in more rigorously designed experiments. For example, much of the case study literature concerning impacts of livestock on riparian wildlife suffers from experimental design inadequacies, including lack of pretreatment data, low sample size, and lack of randomization of treatments (Rinne 1985; Larsen et al. 1996). Additional research based on replicated experimental designs is needed to better understand the relationship between grazing and riparian ecology at scales relevant to determining the ecological consequences of grazing practices.

The second knowledge gap emphasizes ecological processes that mediate the effect of management actions on riparian ecosystem products and services. Without mechanistic understanding of the intervening ecological processes that mediate cause-and-effect relationships, we cannot generalize study results to other sites. This point is particularly important given the dependence of riparian plant species on groundwater resources that vary over both space and time (Stringham et al. 2001; Poole et al. 2006). Improving our knowledge of the effects of management on ecosystem services will involve an expanded, research-based focus on the interaction between management activities and biophysical mechanisms responsible for provisioning ecosystem services.

Several riparian practices have been shown to enhance riparian functions and dependent ecosystem services. (Photo: Ken Tate)

CONCLUSIONS

While the scientific evidence for many riparian management practices is inconclusive, there are several practice benefits that are well documented. There is strong evidence supporting the influence of management

practices on vegetation in riparian habitats, including the following:

1. Grazing intensity influences herbaceous species composition and productivity (H1, 2).
2. Livestock distribution practices, such as water developments, supplement placement, and herding, are effective means of reducing livestock residence time and utilization in the riparian zone (H3).
3. Grazing exclusion can promote recovery of riparian plant community composition in degraded riparian systems (H4).
4. Livestock and other large herbivores can modify the structure and composition of woody plant communities (H5).
5. Late-season (usually late summer and fall) livestock use of riparian areas can lead to increased utilization of woody plants, especially when herbaceous plants are limited in availability or forage quality (H6).

There is also evidence supporting the influence of riparian management practices on riparian vegetation and soils (Fig. 1). Riparian grazing management that maintains or enhances key riparian vegetation attributes (i.e., species composition, root mass and root density, cover, and biomass) will enhance stream channel and riparian soil stability, and this in turn will support ecosystem services, such as flood and pollutant attenuation and quality of riparian habitats (H8).

Finally, limited evidence indicates that riparian habitat management can promote ecosystem services by enhancing vegetation and soil attributes (Fig. 1):

1. Riparian vegetation can function to attenuate waterborne pollutants in overland and flood flow events (H9).
2. The design and implementation of optimally efficient riparian buffers must incorporate site-specific biophysical factors, including buffer width, vegetation attributes and management, pollutant type, pollutant load and concentration, flow rate, hydrologic residence time, and soil attributes (H9).
3. Practices that reduce livestock densities, residence time, and fecal and urine deposition in riparian areas and stream flow generation areas can reduce nutrient and pathogen loading of surface water (H10).
4. Grazing in wetland habitat does not decrease habitat quality for waterfowl except in instances of heavy grazing (H11).
5. Grazing can decrease populations of riparian obligate avifauna but may increase or have no effect on generalist species (12).
6. Shading provided by woody plant cover along with other factors (e.g., elevation, topography, and subsurface flow) will have some effect on stream temperature dynamics (19).
7. Fire can be used to control some species of woody plants, but success of fire-based restoration may be related to availability of water and the availability of propagules of desired species following years of limited water availability (H7).
8. Grazing does not appear to decrease the abundance or overall diversity of macroinvertebrates, but these data are limited (H13).
9. Limited data suggest that livestock grazing practices that are too long in duration and poorly timed can decrease salmonid habitat quality or populations (13).

For several hypotheses, the evidence supporting or refuting beneficial effects on ecosystem services was weak or inconclusive. These include 1) riparian grazing decreases habitat quality for riparian mammals (H14) and sage grouse (H15); 2) woody plant control can reduce undesirable plant species (H16) or increase the abundance of terrestrial wildlife (H17); 3) the influence of riparian burning (H7) on vegetation and animals; 4) prescribed fire can increase the richness, diversity, and abundance of native riparian plants and animals (H 20); 5) upland brush management can decrease erosion and increase stream flow (H 18); and 6) carbon storage can be enhanced by the establishment and maintenance of woody species, herbaceous species with high root mass, and dominance of deep-rooted perennials.

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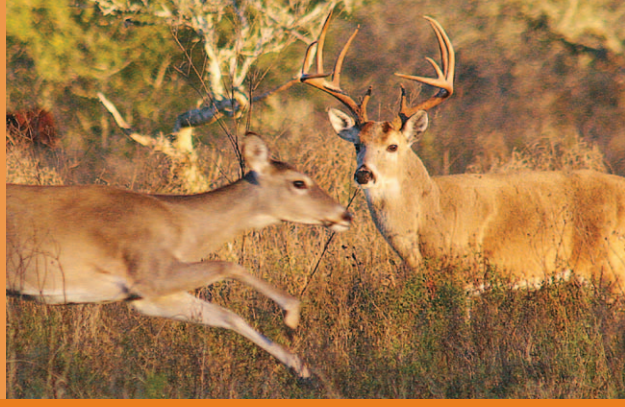

CHAPTER

6

An Assessment of Rangeland Activities on Wildlife Populations and Habitats

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6

Paul R. Krausman, Vernon C. Bleich, William M. Block, David E. Naugle, and Mark C. Wallace

INTRODUCTION

Numerous management practices are applied to rangelands in the western United States to enhance wildlife, including prescribed grazing, burning, brush management, mowing, fencing, land clearing, planting, and restoration to benefit soil and water. Indeed, the Natural Resources Conservation Service (NRCS) lists 167 conservation practices (http://www.NRCS.USDA.gov/technical/standards/nhcp.html). However, wildlife responses to conservation practices are usually species and even species-habitat specific, meaning not only that each species may respond differently to any specific practice but also that a single species may respond differently to the same practice in different vegetation associations or conditions. When managers apply conservation practices to the landscape, habitat is often altered, and managers should understand that the management will benefit some of the wildlife present but may be detrimental to others. Conservation practices were designed to help ecosystem managers think about the variables that accompany any action on the landscape. Each conservation practice has specific purposes that may influence related resource issues. For example, prescribed grazing by large herbivores can alter the structure and function of ecosystems that have direct and indirect effects on wildlife. Primary effects are often described in the literature (Mackie 2000), but there has not been an evaluation of how conservation practices affect wildlife on rangelands. However, practices like prescribed grazing are not a simple treatment but have widely divergent effects, depending on locale, timing, intensity, and species or combination of grazing animals (Briske et al. 2008). Similarly, small mammals, reptiles, amphibians, and bats represent very broad wildlife categories that may have diverse responses to various conservation practices. For example, focusing on Rodentia includes species with such widely different habitat and life history strategies that responses within the group may differ diametrically when exposed to the same management practice. Furthermore, most of the studies that have examined how anthropogenic activities on rangelands influence wildlife have not classified the management activities involved according to the NRCS conservation practices. Thus, we refer to related conservation practices on rangelands that influence wildlife as rangeland activities.

Deer fawn on the Theodore Roosevelt Memorial Ranch, Dupuyer, Montana along the Rocky Mountain Front. (Photo: Sonja Smith)

Wildlife in America has been strongly influenced by agriculture; livestock grazing is the most widespread land management practice in the world (Holechek et al. 2003) and affects 70% of the land surface in the western United States (Fleischner 1994). Traditional practices in rangeland management often homogenize grazing lands to increase forage production and maximize sustainable yield for domestic livestock. New management approaches that promote the spatial and temporal scale of heterogeneity in vegetation structure, composition, and biomass so that sufficient tracks of particular vegetation associations can accommodate desired wildlife populations are needed to improve habitat for wildlife (Fuhlendorf and Engle 2001; Bruno and Cardinale 2008).

The dynamics of native and domestic ungulates, combined with various management practices, create a complex interaction that influences plant and animal communities by altering ecosystem structure, nutrient cycling, productivity, recruitment, predator–

Pyrruhuloxias (*Cardinalis sinuatus*) occupy desert scrub and mesquite-dominated rangelands in southwestern United States. (Photo: Tim Fulbright)

prey relationships, urination and defecation, trampling, and competition. Additional modifications to landscapes, including roads, fences, anthropogenic water sources, agricultural structures, and other developments related to livestock production on western rangelands, modify wildlife behavior and complicate wildlife management. This is especially important for wildlife, as domestic stock and the related anthropogenic developments alter forage availability and cover and contribute to habitat alteration and fragmentation. Large herbivores may potentially modify landscapes in numerous ways (Senft et al. 1987; Ohmart 1996; Fuhlendorf and Engle 2001), but describing them is beyond the scope of this chapter. However, it is not surprising that the effects of prescribed grazing on wildlife have received more attention in the literature than other conservation practices. Many of the early studies of wildlife parallel livestock husbandry and range management theory in that grazing and browsing are the primary factors affecting the kinds, amounts, and quality of forage available (Mackie 2000).

Our objective was to review peer-reviewed literature to examine how conservation practices influence wildlife and wildlife habitats on rangelands in the United States, with specific reference to the NRCS Conservation Practice Standard for Upland Wildlife Habitat Management. The main purpose of this conservation standard is to treat upland wildlife habitat concerns identified during the conservation planning process that enable movement or provide shelter, cover, and food in proper amounts, locations, and times to sustain wild animals that inhabit uplands during a portion of their life cycle. We emphasized the literature compiled in the bibliography by Maderik et al. (2006) but also considered other articles to provide a more complete review.

We documented rangeland activities that influenced (i.e., positive and negative) game birds, nongame birds, carnivores, ungulates, small mammals, reptiles, and amphibians on western rangelands. Carnivores are rarely considered by NRCS, but we include them in our review because of their importance to functioning ecosystems. We also identified gaps in scientific knowledge and recommended future research to enhance management of wildlife on western rangelands in the United States. We supplemented the synthesis with literature outside the United States when similar knowledge within the United States was not available.

RESULTS OF THE LITERATURE ASSESSMENT

Very few of the NRCS conservation practices that directly affect upland wildlife habitat are addressed or evaluated in the peer-reviewed literature. We identified specific activities when appropriate; however, this review is dominated by grazing because of the high profile that grazing has received by the scientific community. Prescribed grazing, when carefully controlled, can be useful in improving habitat for specific species, but the frequency, timing,

and intensity of livestock grazing for maximum wildlife benefits are different than those used for maximum livestock benefits (Holechek et al. 1982). For wildlife, the amount of critical residues left after prescribed grazing is more important than the amount removed; the condition of most ranges will deteriorate when greater than 50% of grazable vegetation is used annually (Hyder 1953; Holechek et al. 1982).

More than 25 yr ago, Holechek et al. (1982) reviewed how prescribed grazing could improve wildlife habitat and concluded that the database was limited. They argued that research into how grazing strategies influence wildlife should receive high priority. Unfortunately, peer-reviewed literature evaluating conservation practices for upland wildlife habitat management, including prescribed grazing, has not received high priority, and the complex influences on wildlife and their habitat remain largely unknown.

Rangeland Activities and Habitat for Game Birds

Conservation practices that improve habitat, if identified and implemented, may halt the decline or, in many cases, enhance the viability of game bird populations. Distribution and abundance of native grouse (subfamily Tetraoninae) that symbolize the biological diversity of western grazing lands are in decline (Knick et al. 2003; Hagen et al. 2004) or are already threatened or endangered (Storch 2007). Exceptions include spruce grouse (*Canachites canadensis* L. 1758) and blue grouse (*Dendragapus obscurus* Say 1823) populations and most white-tailed ptarmigan (*Lagopus leucurus* Richardson 1831) populations. Indigenous quail (subfamily Phasianinae) populations, though stable locally, are largely in decline in the desert Southwest (Saiwana et al. 1998; Western Quail Management Plan 2008) and in the southern Great Plains (Veech 2006). Species considered here that are native to western grazing lands include Gunnison (*Centrocercus minimus* Young et al. 2000) and greater sage-grouse (*C. urophasianus* Bonapart 1827); lesser (*Tympanuchus pallidicinctus* Ridgeway 1873), greater (*T. cupido* L. 1758), and Attwater's prairie-chicken (*T. cupido attwateri*); plains (*T. phasianellus jamesi* L. 1758) and Columbian sharp-tailed grouse (*T. p. Columbians* L. 1758); wild turkey (*Meleagris gallopavo* L. 1758); bobwhite (*Colinus virginianus* L. 1758); and scaled quail (*Callipepla squamata* Vigors 1830).

Our synthesis includes U.S. Department of Agriculture (USDA) Conservation Practices Standards that benefit native grouse and quail and is supplemented with information on exotic species (e.g., ring-necked pheasant [*Phasianus colchicus* L. 1758] and grey or "Hungarian" partridge [*Perdix perdix* L. 1758]) that are abundant regionally and provide recreational and economic benefits (Bangsund et al. 2004). We do not synthesize the rich literature for ring-necked pheasant because in-depth reviews for this species response to Farm Bill conservation practices (Haufler 2007) and other management are readily available (Trautman 1982; Berner 1988; Kimmel and Berner 1998).

We present findings regionally because variation in climatic gradients (Fulbright and Ortega-Santos 2006), disturbance regimes (Coppedge et al. 2008), and contemporary land use change (Foley et al. 2005) influence vegetation response to management. We critically reviewed strength of evidence because variation in study design (Guthery 2007) and ecological scale of investigation (Manzer and Hannon 2005) further influence applicability of research outcomes to management. We placed recommendations within the context of landscape conservation, a well-known ecological principle (Lindenmayer et al. 2008) that is being used in management of game birds at large scales (Hagen et al. 2004; Manzer and Hannon 2005).

Landscape Conservation. Public land managers use holistic strategies that conserve entire landscapes because to be effective the scale at which conservation practices are implemented must match the scale of anthropogenic change that threatens populations. Tillage agriculture (Walker et al. 2007), urban sprawl (Knick et al. 2003; Krausman et al. 2008), tree and shrub invasion (Fuhlendorf et al. 2002), and energy development (Naugle et al. 2011) result in broad-scale loss and degradation of habitat that overwhelms management of remaining fragments (Fuhlendorf et al. 2002; Veech 2006). Wholesale fragmentation increases predation rates (Manzer and Hannon 2005), alters historic disturbance regimes (Baker

> "Very few of the NRCS conservation practices that directly affect upland wildlife habitat are addressed or evaluated in the peer-reviewed literature."

2006), promotes the spread of invasive plants (Bergquist et al. 2007), and facilitates disease (Walker et al. 2007). The concept of conserving the remaining "usable space" is a primary underpinning for quail habitat management in the south-central region of the Great Plains (Guthery 1997). Under this paradigm, managers should strive to increase the quantity of quail habitat. Reversing declines in game bird populations will require regional management of remaining usable space (Williams et al. 2004).

Rangeland Activities. Most literature documents the decline or extirpation of wildlife populations that result from chronic overgrazing. Overgrazing is defined here as the combination of stocking rates and timing of grazing that reduces wildlife reproduction and survival by altering the short- and long-term structure and composition of grassland and shrubland vegetation. This chapter may be frustrating for some readers looking for precise guidance because little experimental research has been conducted to know which conservation practices benefit game birds. Most contemporary studies lack experimental controls, are too short in duration, and fail to collect pretreatment data. Moreover, findings cannot be readily translated in conservation practices (e.g., prescribed grazing) because existing studies typically compare wildlife response to grazed and ungrazed pastures without reference to grazing strategy, regime, or system. Implications should not be extrapolated too broadly because they are most often derived from studies of specific species and local-scale management actions.

Grazing. Livestock grazing is a controversial practice because indirect evidence overwhelmingly suggests that overgrazing reduces nest success (e.g., scaled quail [Pleasant et al. 2006], ring-necked pheasant [Clark and Bogenschutz 1999] and greater sage-grouse [Beck and Mitchell 2000]) and brood survival (lesser prairie-chicken [Hagen et al. 2005] and wild turkey [Spears et al. 2007]) by decreasing height and density of herbaceous cover. Livestock grazing can have negative or positive impacts on game bird habitat, depending on timing and intensity of grazing and which habitat component is being influenced (Beck and Mitchell 2000). Light to moderate grazing can promote forb abundance (e.g., food), but heavy grazing reduces herbaceous cover and promotes invasive species (Crawford et al. 2004). Guidelines describing height and density of herbaceous cover necessary to maintain productive habitats are available for many game bird species (Connelly et al. 2000; Hagen et al. 2004). These guidelines provide the "biological sideboards" necessary to guide grazing strategies for maintaining and enhancing populations; unfortunately, the grazing strategies necessary to achieve the necessary cover requirements for game birds are poorly understood.

The only empirical evidence of the influence of prescribed grazing on game birds we found in the literature was an unpublished report (Rice and Carter 1982) from a 5-yr study of game birds at Fort Pierre National Grassland in central South Dakota. Authors compared deferred rotation, rest–rotation, and winter-only grazing. Pastures (404 ha) that were deferred from grazing until winter provided the highest number of plains sharp-tailed grouse and greater prairie chicken nests and broods. Rest–rotation grazing accommodated the second-highest density of nests and broods for both species. Deferred rotation did not provide blocks of undisturbed cover available in the spring for nesting, which was reflected in the lowest density of nests and broods. Pastures managed under rest–rotation grazing, which had the highest cattle stocking rate of any system, produced approximately 10 times more nest-broods than did pastures managed in a deferred rotation system. During the 5-yr study, grouse followed the grazing rotation seeking the best herbaceous cover for nesting and rearing broods. Grouse preferred rested pastures for nesting that were at times 4.0 km from breeding sites.

In the south-central United States (e.g., Texas and Oklahoma), grazing management can be prescribed to benefit bobwhite habitat, but a large part of potential quail range in the Rolling Plains has been overgrazed and excessively treated for brush control (Rollins 2007). Today, more landowners are tempering traditional land management goals with more quail-friendly practices, including reduced stocking rates (Rollins 2007). Adequate nesting cover is a key consideration for quail managers

Sage-grouse (*Centrocercus urophasianus*) are an important species of concern in western rangelands. (Photo: Brett Billings)

(Slater et al. 2001) because food is rarely the limiting factor for bobwhites in Texas (Guthery 2000). Livestock grazing can be an effective tool for managing quail habitat, especially in manipulating plant succession (Guthery 1986). But across most of Texas, bobwhite abundance declines as cattle density increases (Lusk et al. 2002). Light to moderate stocking rates that provide 50% grass and 20% to 30% woody vegetation result in adequate bobwhite nesting habitat in western Oklahoma (Townsend et al. 2001). Guthery (1986) emphasizes flexibility in grazing prescriptions to allow "slack" (Guthery 1999) in the system to account for variability in brush cover and short- and long-term precipitation patterns.

Other than the examples mentioned above, little experimental data are available to identify beneficial grazing practices that increase bird populations levels (e.g., greater sage-grouse [Connelly et al. 2000] and lesser prairie-chicken [Pitman et al. 2005]) because mechanisms are poorly understood (Beck and Mitchell 2000; Hagen et al. 2004). Effects of livestock grazing vary regionally because, unlike the Great Plains where bison (*Bos bison* H. Smith 1827) once flourished (Sanderson et al. 2008), many semiarid sagebrush and arid desert ecosystems evolved with substantially less grazing (Connelly et al. 2000; Knick et al. 2003). Wildlife managers in the Great Plains readily acknowledge the importance of livestock grazing to conservation because ranchers whose operations remain profitable are less likely to convert native prairie to cropland (Licht 1997; Higgins et al. 2002). Conversely, wildlife managers in sagebrush and desert grasslands see grazing as detrimental because excessive stocking rates often results in severe habitat degradation (Mack and Thompson 1982; Knick et al. 2003). We need more experimental studies like those in Europe showing how managed grazing was used to recover a declining population of black grouse (*Tetrao tetrix* L. 1758) in northern England (Calladine et al. 2002). Black grouse numbers averaged 6.3% higher per year, and brood survival was 22% higher at sites with reduced grazing than in overgrazed reference sites.

Vegetation Manipulations Detrimental to Populations. A host of vegetation manipulations that detrimentally impact game birds include agricultural tillage, herbicide application, mechanical sagebrush removal, and overprescription of fire in xeric landscapes. Tillage agriculture directly reduces the amount of habitat available and fragments remaining grasslands to the detriment of wildlife populations (Swenson et al. 1987). Various means of mechanical and herbicidal removal of sagebrush (*Artemisia* spp.) directly reduce the abundance of shrub and herbaceous vegetation that sage-grouse rely on for food and cover (Wallestad 1975; Braun and Beck 1996). Periodic fire may rejuvenate grasslands in the Great Plains (Reinking 2005; but see Patten et al. 2007), but widespread burning of sagebrush landscapes is not warranted in xeric environments farther west (Beck et al. 2008). Similarly, lesser prairie-chickens in southeastern New Mexico shrublands selected sand shinnery oak (*Quercus harardii* Rydb.) landscapes for thermal refugia and protective overhead cover; selection for these landscapes suggests no justification for shrub control for prairie-chicken conservation in these landscapes (Bell et al. 2010).

Western populations of painted buntings (*Passerina ciris*) breed in the shrublands of northern Mexico and Texas. (Photo: Tim Fulbright)

“A science-based approach is the key to implementing the right practices in the right places and then documenting outcomes to wildlife populations.”

Exotic and Woody Plant Invasions. Activities that enable proliferation of exotic herbaceous and woody plants (e.g., tree/shrub establishment) in rangelands should be avoided, but those that reduce or remove unwanted invasive species are encouraged (Flanders et al. 2006). Game bird populations have suffered from human fire suppression that promotes tree and shrub invasions and establishment of exotic plants that eventually results in catastrophic wildfire. An increase in tree abundance is associated with lower persistence of lesser prairie-chicken populations in Oklahoma and Texas (Fuhlendorf et al. 2002). Sage-grouse do not use mountain big sagebrush (*Artemisia tridentata* Nutt. *vaseyama* [Rybd.] Bettle) landscapes that are invaded by pinyon (*Pinus* spp.)–juniper (*Juniperus* spp.) woodlands at higher elevations in the intermountain West (Miller et al. 2000; Crawford et al. 2004); the exact mechanism is unknown, but birds either experience higher predation rates or avoid tall structures in otherwise suitable habitats. Similary, scaled quail avoid grasslands invaded by trees in the desert Southwest (Van Auken 2000; Bristow and Ockenfels 2006). Another major problem throughout much of the West is proliferation of cheatgrass (*Bromus tectorum* L.), which reduces viability of game bird populations. Invasion of rangelands by cheatgrass has led to a cycle in which increasing abundance of this annual grass promotes large fires that allow cheatgrass to increase further, causing the loss of perennial bunchgrasses and low-elevation communities of Wyoming big sagebrush (*Artemisia tridentata wyomingensis* Beetle and Young) (Knick 1999; Baker 2006). This phenomenon is particularly troubling because no large-scale restoration techniques are currently available to restore the millions of ha of sagebrush-dominated rangelands that have been lost to wildfire.

Brush Management. South-central Great Plains rangelands have changed greatly over the past century as mesquite (*Prosopis glandulosa* Torr.) savannas become increasingly dense because of a lack of prescribed fire and regrowth from chemical and mechanical brush management. Light to moderate stocking rates usually provide the proper proportions of bare ground, herbaceous quail foods, and woody cover required to sustain bobwhite populations in Oklahoma (Townsend et al. 2001). Grazing intensity will range relative to how much brush is present; lighter stocking rates are required to maintain more herbaceous cover if little brush is present, but heavier stocking rates are possible if more brush canopy is present (Guthery 2002). In the Rolling Plains of Texas, bobwhites selected rangelands containing higher brush canopy cover and overall visual obstruction over those with more bare ground (Ransom et al. 2008). Weather has a tremendous influence on the amount of cattle forage available, leading Lusk et al. (2007) to conclude that reducing livestock stocking rates during dry periods likely will foster ground cover more similar to that available during wet periods. The main factors influencing bobwhite numbers in southern Texas were rainfall during the previous growing season and type of range, with treatments to reduce brush only nominally affecting bird abundance (Cooper et al. 2009). In the same areas of Texas, application of prescribed fire at large spatial scales was deemed a neutral practice for managing bobwhite habitat in semiarid rangelands (Ransom and Schulz 2007).

Strategic Approach to Implementing Beneficial Practices. Implementing practices that are beneficial to game birds is often challenging because many of the critical experiments have not been done to document positive population responses to management. A science-based approach is the key to implementing the right practices in the right places and then documenting outcomes to populations to identify and replicate our successes, manage adaptively to improve delivery, and provide accountability to all our audiences. Implementation of conservation practices should be linked with field-based experimental research to identify the most effective and least expensive ways to benefit wildlife populations. Many birds use habitats at a spatial scale that is larger than that of an individual pasture or ranch. Therefore, our scientific assessments should reflect appropriately large scales at which game bird populations use habitat resources year-round and transcend that of an individual ranch to encompass multiple and nearby ranches enrolled in conservation programs.

The USDA is trying new and innovative ways to link science with implementation to document the benefits of NRCS conservation

practices. For example, the USDA launched its new and exciting Sage-grouse Initiative (SGI) in March 2010 to provide a holistic approach to conserving sage-grouse and sustaining working ranches in the West. In its inaugural year, the SGI has quickly become one of the largest and most recent conservation success stories in the West. The SGI's success is in capitalizing on the strong link between conditions required to support sustainable ranching operations and habitats that support healthy sage-grouse populations. The SGI is a science-based initiative with evaluations carried out by reputable, independent scientists to measure the biological response of sage-grouse populations to conservation practices, to assess SGI effectiveness, and to adaptively improve program delivery.

The SGI follows three primary steps in evaluating the benefits of conservation practices that may serve as a model for others dealing with uncertainty in their implementation effectiveness. First, the NRCS worked with the Bureau of Land Management to map rangewide sage-grouse population centers, or "core areas," to refine SGI delivery ensuring that practices benefit large numbers of birds (Doherty et al. 2010). Targeting practices within core areas ensures that enough of the right conservation practices are implemented in the right locations to anticipate a positive population response. Similar guidance is emerging for targeting conservation practices to benefit sustainable bobwhite quail populations in the West Gulf Coastal Plain (Twedt et al. 2007). Second, SGI-sponsored studies are under way in six states across the West to assess benefits of grazing systems and removal of encroached conifer. Assessments incorporate before–after control–impact designs using radio-marked birds across appropriately large time and space scales to quantify the biological and population-level response of birds to conservation practices. Third, the NRCS completed a conference report with the U.S. Fish and Wildlife Service (USFWS) that proactively amends a suite of 40 conservation practices to ensure they are either benign or beneficial to sage-grouse, including upland habitat management, prescribed grazing, and brush management for juniper removal. By conditioning NRCS conservation practices, private landowners enrolled in SGI can rest assured that they can continue normal ranching operations even if USFWS lists sage-grouse as a federally threatened or endangered species. Collectively, these three steps offer an approach for implementing conservation practices while documenting their success and adaptively improving them when necessary.

Rangeland Activities That Improve Habitat for Nongame Birds

Rangeland management has great potential to improve nongame bird habitat (Haufler and Ganguli 2007). To date, most studies address management effects, not necessarily benefits, on focal species or avian communities. This is logical because biologists must first understand the nature of the effects (e.g., positive, negative, or neutral) to effectively use a given management practice as a tool. However, the science has not progressed much beyond this preliminary phase, and experimental studies designed specifically to evaluate management actions to benefit wildlife are rare. We approach the review of conservation practices to improve habitat for nongame birds with a brief mention of key effects papers and then review papers that evaluate the efficacy of management with the primary objective to improve nongame bird habitat.

Effects Papers. By far, the focus of most research has been to address effects of livestock grazing on nongame birds (Fleischner 1994; Saab et al. 1995; Zimmerman 1997). Research

Young white-tailed deer (*Odocoileus virginianus*) in eastern Wyoming. (Photo: David Briske)

Pronghorn antelope (*Antilocapra americana*) on the Charles M. Russell National Wildlife Refuge, Montana. (Photo: Jeffrey Wright)

has been conducted also to understand effects of fire, mowing, and exotic flora and fauna (Herkert et al. 1996; Zimmerman 1997; Askins et al. 2007). Effects are attributed primarily to changes in habitat structure and composition (Bock and Webb 1984), although trampling of ground nests occur. Indirect effects are ascribed to changes in ecosystem structure that can influence ecological relationships among species. The focus of much attention here concerns brown-headed cowbirds (*Molothrus ater* Boddaert 1783) that parasitize nests of many cup-nesting species.

Given that the focus of this chapter is not to review these studies, suffice it to say that effects on species vary from positive to negative. Perhaps the relevance of these effects studies is that they indicate management activities that are benign, beneficial, or detrimental to species, which is a critical first step in developing proactive management prescriptions.

Grazing. Various studies evaluate grazing as a tool to enhance nongame bird habitat. Grazing is not restricted to exotic domestic herbivores but also includes native species, such as bison and elk (*Cervus canadensis* L. 1758). Indeed, many, if not most, ecosystems rely on grazing by native ungulates to influence vegetation structure and composition (Stebbins 1981); thus, some form and level of grazing may be compatible with natural ecosystems processes. Grazing variables that can be manipulated to achieve nongame bird goals include stocking rates, seasonality, duration, and livestock species. A premise of prescribed grazing is that, if done correctly, it will enhance horizontal heterogeneity and provide a mosaic of landscape conditions to meet a wide range of bird preferences (Herkert et al. 1996; Derner et al. 2009).

Wetland Birds. Grazing improved habitat for wading birds in Austria (Kohler and Rauer 1991). Two factors led to degraded habitat: conversion of pastureland to agriculture and the cessation of grazing that allowed for encroachment of common reeds (*Phragmites* spp.) and rushes (*Juncus* spp.) into pastureland. Cattle were introduced to control the encroachment of reeds and rushes, but Kohler and Rauer (1991) noted no tangible increases of wading bird populations. Tichet et al. (2005) evaluated grazing regimes (stocking levels and seasonality) on use by wading birds in French wetlands. They found that grazing intensity affected species responses differently, depending on their habitat requirements. Curlews (*Numenius arquarta* L. 1758) used areas with greater spring grazing intensity, whereas redshank (*Tringa tetanus* L. 1758) occupancy declined. In autumn, lapwings (*Vanellus vanellus* L. 1758) showed a positive relation to grazing, whereas responses by black-tailed godwits (*Limosa limosa* L. 1758) were negative.

Grassland Birds. Paine et al. (1997) compared three grazing regimes in Wisconsin: grass farms, continuously grazed pastures, and "bird-friendly" rotational systems whereby grazing was deferred to create nesting refuges during the breeding season. They reported that refuges attracted 11% more nesting birds than grass farms and that grass farms attracted 65% more nesting birds than continuously grazed pastures. Nest success for grass farms ranged from 6% to 24% and from 30% to 39% for refuges and was 25% for continuous grazing during both years of study. Most nest mortalities for grass farms and refuges were from mowing. Overall avian productivity within refuges was greater than that for grass farms, which were greater than continuously grazed pastures. Productivity is defined as the number of birds fledged from nests. Temple

et al. (1999) also compared grazing regimes in Wisconsin and reported that diversity, density, nest success, and productivity of grassland birds was greatest on ungrazed lands. Continuously grazed pastures had the lowest diversity and densities but were intermediate for nest success and productivity. Rotationally grazed pastures had intermediate diversity and densities but the lowest nest success and productivity. They recommended a mosaic of ungrazed and rotationally grazed areas to increase productivity of grassland birds above that found on a mosaic of continuously and rotationally grazed pastures (Temple et al. 1999).

Derner et al. (2009) suggested that livestock could be used as "ecosystem engineers" to modify vegetation structure within and among pastures and provide for habitat needs of grassland birds of the Great Plains. Grazing is often used in combination with patch burning to provide the desired vegetation structure. For example, localized grazing and fire could be used to reduce vegetation cover and provide feeding sites for mountain plover (*Charadrius montanus* Townsend 1853) or nest sites for the long-billed curlew (*Numenius americanus* Bechstein 1812). For the upland sandpiper (*Bartramia longicauda* Bechstein 1812), reduced grazing could be used to provide tall vegetation required for nesting, whereas more intensive grazing could increase food availability and enhance foraging habitat.

Riparian Birds. Livestock grazing can have positive and negative effects on habitats for different species of birds riparian systems. Although grazing removes lower vegetation layers, it also influences seedling establishment and regeneration of shrubs and trees. Indeed, dramatic changes in vegetation structure can be seen shortly after livestock are removed from riparian areas (Krueper et al. 2003). In the Northwest, vegetation recovery following livestock removal in a riparian meadow was complex, given interactions with precipitation (Dobkin et al. 1998). Cattle removal resulted in a more diverse and abundant avian community that was even greater in wet years than in dry years. The northern harrier (*Circus cyaneus* L. 1766), common snipe (*Gallinago gallinago* L. 1758), short-eared owl (*Asio flammeus* Pantoppidan 1763), song sparrow (*Melospiza melodia* Wilson 1810), and yellow-headed blackbird (*Xanthocephalus xanthocephalus* Bonaparte 1826) were found only within the cattle-excluded area. In southeastern Arizona, density of herbaceous vegetation increased four- to sixfold following removal of cattle (Krueper et al. 2003). Mean numbers of detections during bird surveys increased for 42 species (26 significantly) and decreased for 19 species (8 significantly) 3 yr following the removal of cattle. Number of individuals detected per kilometer more than doubled. Detections of open cup-nesting species increased the most and Neotropical migratory birds more than others.

Brown-Headed Cowbird Control. Reductions in cattle stocking by 86% (752 animal units [AUM] to 103) were made to decrease nest parasitism on the endangered black-capped vireo (*Vireo atricapilla* Woodhouse 1852) in Texas (Kostecke et al. 2003). Rates of cowbird parasitism decreased by 13 times after cattle were removed. Further, cowbirds needed to travel further to breed, resulting in greater energetic costs and reductions in numbers of eggs laid. There was no evidence of cowbird nest parasitism following removal of cattle from a riparian area in southeastern Oregon, even though nest parasitism was prevalent in nearby riparian habitats where cattle remained (Dobkin et al. 1998).

Multiple Range Activities. Walk and Warner (2000) compared burned, mowed, hayed, grazed, and undisturbed management regimes on areas of introduced cool-season grasses, native warm-season grasses, and annual forbs. Eastern meadowlark (*Sturnella magna* L. 1758) and dickcissels (*Spiza americana* Gmelin 1789) were detected most often among grazed warm-season grasses. Henlow's sparrows (*Ammodramus henslowii* Audubon 1829) and field sparrows (*Spizella pusilla* Wilson 1810) were detected more often among undisturbed warm-season grasses where eastern meadowlarks and grasshopper sparrows (*Ammodramus savannarum* Gmelin 1789) were least abundant. Grasshopper sparrows were most abundant among annual weeds where Henlow's sparrows and field sparrows were not observed. Overall abundance was least among recently burned cool-season grasses. Low-intensity late-season grazing was

Mule deer (*Odocoileus hemionus*) on the Theodore Roosevelt Memorial Ranch, Dupuyer, Montana. (Photo: Sonja Smith)

> …cattle removal or reduction seems to be an effective tool to reduce brown-headed cowbird numbers and nest parasitism on open cup-nesting birds."

important for creating a heterogeneous mosaic to accommodate many of the grassland birds studied.

Griebel et al. (1998) evaluated bird use of two different grazing treatments: 1) bison grazing (year-round; 1.2 AUM · ha^{-1} · yr^{-1}) combined with prescribed fire and 2) cattle grazing (15 May–15 November; 1.0 AUM · ha^{-1} · yr^{-1}). Few differences were reported in bird species richness or relative abundance of species between grazing treatments, vegetation density, and height. During 1 of the 2 yr of study, bird species richness was greater in the bison-fire enclosure than in the cattle enclosure; abundances of lark sparrows (*Chondestes grammacus* Say 1823) and mourning doves (*Zenaida macroura* L. 1758) were higher and grasshopper sparrow lower in bison-fire enclosures. Within the bison-fire enclosures, differences existed between burned and unburned transects, with grasshopper sparrow abundance higher in unburned areas and mourning dove and lark sparrow abundances higher in burned areas.

Danley et al. (2004) reported few differences in bird species diversity or abundance between areas that were burned and grazed versus areas only burned in North Dakota. The notable exception was the brown-headed cowbird, which occurred 2.4 times more frequently on burned and grazed plots.

LaPointe et al. (2003) evaluated use of a rest–rotation grazing system targeted to improve plant cover for nesting ducks and grassland birds along the St. Lawrence River, Quebec. They evaluated four methods: cattle removal, grazing augmented with seeding of forage plants, seeding with no grazing, and seasonal grazing after duck nesting. Overall abundance of birds exhibited no change 2 yr posttreatment. However, bobolink (*Dolichonyx oryzivorus* L. 1758) were more abundant in areas that were seeded with no grazing and where cattle grazed after ducks had nested, and red-winged blackbirds (*Agelaius phoeniceus* L. 1756) were more abundant in the two treatments with no grazing.

Rangeland Restoration. Ecological restoration is a management paradigm whose objective is to return conditions to those that existed in the past, typically those that occurred prior to European settlement of North America. Implicit is that, in doing so, avian community structure and composition will be restored also. At this point, results of the few studies that have evaluated effects of restoration of birds are equivocal.

Fletcher and Koford (2003) evaluated effects of restoring native grasslands from former agricultural (e.g., hay land and row crops) land and reported that 16 of 54 species detected increased with restoration. Only killdeer (*Charadrius vociferus* L. 1758) and cowbird responded negatively to restoration. Five of the species that increased are of broad regional concern because populations are declining. In contrast, Van Dyke et al. (2004) found no bird responses, positive or negative, to the use of fire and mowing to restore tallgrass prairie in Iowa. Results may have been influenced by the small scale (< 10 ha) of treatments.

In southeastern Arizona, Malcolm and Radke (2008) evaluated effects of active wetland and riparian restoration following passive restoration (e.g., cattle removal) on bird density and diversity. Cattle removal occurred in 1980 and was followed by active restoration in 2005. Active restoration consisted of installation of erosion control gabions to create two wetlands that were then used to irrigate a desert scrub plot. Bird densities increased by 2.3 birds · ha^{-1} in 2006 and 8.4 · ha^{-1} in 2007 following active restoration treatments. Species richness showed a marginal difference.

Kennedy et al. (2008) compared cover by native versus nonnative plants and the resulting influence on nest productivity of passerine birds. They reported no association between the percentage of nonnative plant cover and nest densities, clutch size, productivity, nest survival, and nestling size.

Overall, studies evaluating effects of range management directed at improving nongame bird habitat are rare. Many studies are essentially case studies whose results apply largely to the place and time of study. As a result, generalizations are difficult at best. Some trends that emerged from the papers reviewed are that continuously grazed pastures

appear to have fewer birds and fewer species than areas grazed using a rotational system, grazed after the breeding season, or where cattle were removed entirely. Additionally, cattle removal or reduction seems to be an effective tool to reduce brown-headed cowbird numbers and nest parasitism on open cup-nesting birds.

Rangeland Activities and Habitat for Carnivores

References regarding influences of rangeland activities on carnivores are notably sparse and are rarely considered by the NRCS. However, we include them in this review because of their importance to functioning ecosystems. We considered 14 taxa to be representative of western rangeland habitats: coyote (*Canis latrans* Say 1823), wolf (*Canis lupus* L. 1758), kit fox (*Vulpes macrotis* Merriam 1888), swift fox (*Vulpes velox* Say 1823), red fox (*Vulpes vulpes* L. 1758), grey fox (*Urocyon cinereoargenteus* Schreber 1775), black bear (*Ursus americanus* Pallos 1780), grizzly bear (*Ursus arctos horribilis* L. 1758), mountain lion (*Puma concolor* L. 1771), bobcat (*Lynx rufus* Schreber 1777), raccoon (*Procyon lotor* L. 1758), striped skunk (*Mephitis mephitis* Schreber 1776), spotted skunk (*Spilogale* spp.), and black-footed ferret (*Mustela negripes* Audubon and Bachman 1851).

Based on references in the bibliography of Maderik et al. (2006), rangeland activities appear to influence habitat for spotted skunks and striped skunks (Neiswenter and Dowler 2007). Spotted skunks use areas with more large mesquites than striped skunks, and striped skunks did not select any habitat relative to its availability, but both species appeared to avoid agricultural areas. Conservation of western spotted skunks may be enhanced by limiting brush control for management of livestock on mesquite dominated rangelands (Neiswenter and Dowler 2007).

Others reported that the distribution and shape of grassland patches, woodland patches, pastureland, and farmsteads influenced detections of striped skunks, raccoons, and red fox. Kuehl and Clark (2002) determined that evidence of striped skunks decreased as distance from grassland patches increased but, in contrast to Neiswenter and Dowler (2007), was positively associated with the number of farmsteads in their study area. Raccoon presence was positively related to presence of woody cover, and red fox presence increased with greater area of pastureland and greater isolation from farmsteads but decreased with increasing amounts of habitat arranged in strips across the landscape. Ivan et al. (2002) reported that alteration of prairie landscapes through increases in planted trees, woody cover, rock piles, and junk piles enhanced conditions for striped skunks and raccoons by providing denning habitat. Maestas et al. (2003) concluded that ranchlands supported relatively more coyotes than exurban developments and that ranches are important for protecting biodiversity, suggesting that future conservation efforts may require less reliance on reserves and a greater focus on private lands. In ecologically similar areas of Arizona, Horejsi (1982) reported that coyotes were relatively more abundant on ungrazed than on grazed rangelands; however, the ungrazed area had been closed to predator control for an extended period of time prior to the initiation of his research, and the other had not. These results have implications for predicting the influence of rangeland management practices on specific species of carnivores and, through their affects on

Rio Grande wild turkeys (*Meleagris gallopavo intermedia*) are found along riparian areas and in shrublands. (Photo: Tim Fulbright)

> “…influences of rangeland activities on large native carnivores have nearly all been negative.”

landscape configuration, on conservation of biodiversity in general.

Hilty and Merenlender (2004) emphasized that wide, well-vegetated riparian corridors are important in maintaining the connectivity of native predator populations to ensure their long-term survival. In a similar riparian system, Ammon and Stacey (1997) concluded that livestock grazing reduced streamside vegetation and that grazing could influence predator assemblages and, thereby, affect bird populations directly and indirectly. Cattle grazing did not affect vegetation height or density along edges of pasturelands compared to the interior of pasturelands. Raccoons and other predators may move more freely in pasturelands when compared with edges of pasturelands, thereby explaining an absence of differences in predation risk for nesting grassland birds in those habitats (Renfrew et al. 2005). Conversion of rangelands to irrigated agriculture (i.e., alfalfa, mint, and sugar beets) may have a positive effect on burrowing owls (*Athene cunicularia* Molina 1782) where those small strigids use burrows abandoned by badgers (*Taxidea taxus* Schreker 1777; Belthoff and King 2002); presumably, such practices have a negative affect on badgers although not explicitly stated.

Numerous references included in the bibliography by Maderik et al. (2006) (Beck and Mitchell 2000; Townsend et al. 2001; Herkert et al. 2003; Cox et al. 2005; Miller and Guthery 2005; Renfrew et al. 2005; Shochat et al. 2005; Sutter and Ritchison 2005; Grant et al. 2006) make inferences about onerous affects of grazing on predator assemblages or the ability of predators in general to better detect and prey on the nests of ground-nesting birds. Results of these investigations address primarily changes in predation risk to ground-nesting birds as a result of modifications to habitat structure or composition rather than changes in carnivore populations themselves.

Generalizations About Overall Effects of Management on Carnivores

Habitat alteration and loss and harvesting for sustenance, sport, and profit have resulted in substantial declines in top predators in a wide variety of habitats (Bruno and Cardinale 2008), including rangelands of western North America (Laliberte and Ripple 2004). Overgrazing of rangelands by domestic livestock, sometimes combined with other practices, has influenced the structure and composition of rangeland habitats, with resultant impacts to biodiversity and ecosystem function (Blaum et al. 2007). Additionally, efforts to enhance livestock production on those rangelands have included attempts to eliminate carnivores viewed largely as predators of livestock. As a result, influences of rangeland activities on large native carnivores have nearly all been negative. Nevertheless, some medium-sized carnivores (e.g., coyotes, skunks, and raccoons) have experienced increases in populations and distribution, in part resulting from an enhanced food base associated with human presence or the absence of predators that no longer compete with or prey on those carnivores.

Four of the taxa (i.e., wolf, grizzly bear, black-footed ferret, and San Joaquin kit fox [*Vulpes macrotis mutica* Merriam 1902]) have been impacted by activities associated with rangeland management (i.e., predator control activities, habitat modification, and conversion) to the extent that they have been afforded federal protection under the Endangered Species Act. Two others (i.e., mountain lion and swift fox) have suffered substantial reductions in distribution and numbers.

Throughout much of the history of western North America, ranchers and other livestock producers have viewed large carnivores as incompatible with production objectives. Ranchers and other rangeland managers viewed predator management as an augmentation of the efficacy of other practices, and, as such, it has become a widespread and accepted practice throughout much of the United States. Although predator control is not explicitly one of the NRCS rangeland management practices currently in place, it has been (and in some cases likely will continue to be) an activity that occurs in conjunction with current NRCS practices that place an emphasis on habitat quality and enhancement. As such, a brief history of predator management and its impacts on species and ecosystems is warranted in this chapter. Moreover, some carnivores have benefited from implementation of selected NRCS management practices and warrant recognition.

Widespread efforts to eliminate wolves and grizzly bears from rangelands in the 48 contiguous states were largely successful (Young 1944; Storer and Tevis 1955; Mech 1970; Brown 1985), and the use of a variety of techniques, including widespread campaigns of poisoning, trapping, and shooting, ultimately resulted in the previously mentioned classification of those large carnivores as endangered taxa. Another large carnivore, the mountain lion, also was the object of less successful but still intensive (Bruce 1953; Hert and McMillin 1955) efforts to reduce impacts to livestock operations.

Gray wolves once ranged throughout much of North America but were systematically eliminated from the majority of historical habitats in part because of the threat to livestock (Musiani and Paquet 2004). Indeed, it is estimated that wolves had been eliminated from greater than 85% of their former range in rangeland habitats prior to restoration efforts (Laliberte and Ripple 2004). Nevertheless, federal protection, combined with efforts to manage wolves in the north-central United States (Mech 1970) and efforts to restore them within historical ranges in the northern Rocky Mountains (USFWS 1987), has been successful. Wolves remain important predators of livestock, but current management strategies include provisions for removal of offending individuals.

Grizzly bears once occupied suitable habitat across a wide expanse of the continental United States, but their geographic range has been reduced by 91% in temperate grasslands, savannas, and shrublands and by 100% in desert and xeric shrublands (Laliberte and Ripple 2004), largely a result of efforts to eliminate historic conflicts with livestock grazing and other human activities. Grizzly bears were afforded protection under the Endangered Species Act in 1975, and an initial recovery plan was completed in 1982 and revised in 1993 (USFWS 1982, 1993). Currently, grizzly bears are categorized as 1) an experimental, nonessential population segment in parts of Idaho and Montana and 2) a recovered distinct population segment in the Greater Yellowstone Ecosystem of Idaho, Montana, and Wyoming. Elsewhere in the continental United States, grizzly bears remain listed as threatened, but the status of populations inhabiting the Cabinet-Yaak Recovery Zone, the Selkirk Recovery Zone, and the North Cascades Ecosystem Recovery Zone are under review (USFWS 2008). Recovery of grizzly bears is dependent on the maintenance of suitable habitat in occupied areas and judicious management of individuals that prey on livestock.

Populations of swift foxes and kit foxes declined substantially as a result of rangeland activities, and their influences, including habitat loss through conversion of native prairies, trapping, predator control, shooting, collisions, and use of rodenticides to control prey populations, likely contributed to the decline of swift foxes (Carbyn 1995; Meaney et al. 2006). Further, unanticipated trophic cascades due to widespread removal of wolves and subsequent increases in coyotes and, potentially, red foxes, which prey on or compete with these small canids, likely have contributed to the decline of swift and kit foxes (Carbyn 1995; Cypher et al. 2001; Meaney et al. 2006). Alteration of native prairies due to grazing and agricultural practices has been especially problematic for these foxes, and losses were exacerbated by poisoning, trapping, and other efforts to manage larger predators, including coyotes and wolves (USFWS 1983, 1995).

Mountain lions can be important predators of livestock, particularly domestic sheep, which are grazed widely on western rangelands. Efforts to reduce mountain lion populations were intense during the early 20th century (Bruce 1953; Hert and McMillin 1955), but those activities declined substantially in most of the western states by the 1970s. Nevertheless, it is estimated that the geographic range of mountain lions occupying western rangelands has been reduced by 49%; distribution of those large felids in desert and xeric shrublands has, however, remained unchanged (Laliberte and Ripple 2004). Although mountain lions were successfully eliminated from a substantial proportion of their historical distribution, they remain the most widely distributed large carnivore in North America (Pierce and Bleich 2003). In some areas of the southwestern United States, mountain lion populations have been subsidized by increased food supplies in the form

of domestic livestock that allow mountain lions to persist at higher densities, and, as a result, the effects of predation on native ungulates have been exacerbated (Rominger et al. 2004). Increased shrub cover on rangelands often is associated with overgrazing (Blaum et al. 2004), with resultant influences on biodiversity of mammalian carnivores (Blaum et al. 2007) that may enhance hunting efficiency of mountain lions. Reduction of shrub cover on rangelands may decrease hunting efficiency of mountain lions, and conversion of cow–calf operations to steer operations may decrease the benefits of livestock operations to mountain lions and, thereby, reduce their impacts on native ungulates (Rominger et al. 2004). Currently, mountain lions are managed as a game species in the majority of western states, but exceptions occur (Pierce and Bleich 2003; Bleich and Pierce 2005).

Black-footed ferrets have declined substantially in distribution and once were thought to be extinct in the wild. Widespread poisoning campaigns to eliminate prairie dogs (*Cynomys* spp., a principal prey of these endangered mustelids) from rangelands were implicated in the near extinction of that species, as has conversion of rangeland to cropland (USFWS 1988). As a result of a captive breeding program, black-footed ferrets have been translocated to appropriate habitats in several states but remain one of the most critically endangered mammals in North America.

Coyotes have been an unanticipated beneficiary of widespread efforts to reduce wolves, and the distribution and range of coyotes have increased substantially as a result. Although direct mortality of coyotes due to wolf predation was low, results of recent research are consistent with the hypothesis that coyote abundance is limited by competition with wolves (Berger and Gese 2007). Trophic cascades involving wolf removal and resultant expansion of the distribution of coyotes, a generalist predator, have resulted in further impacts to smaller canids, including swift fox and kit fox (Cypher et al. 2001). Coyote control is an important rangeland activity, and substantial research on control efficacy and methodology has been conducted (Knowlton et al. 1985, 1999; Shivik 2006). Coyotes remain an important predator of livestock, particularly domestic sheep, but government-subsidized predator control alone has failed to prevent a decline of the sheep industry (Berger 2006). Coyote control to benefit livestock production can, however, have a positive effect on native ungulates, including mule deer (*Odocoileus hemionus* Rafinesque 1817) and pronghorn (*Antilocapra americana* Ord 1815; Harrington and Conover 2007). Similarly, interference competition by wolves with coyotes has a positive influence on survival of pronghorn fawns (Berger et al. 2008).

In general, mammalian carnivores have benefited little from rangeland management activities. An exception is the coyote, a generalist predator that has expanded its distribution substantially as a result of the extirpation of the wolf from the majority of its historical range. Such shifts have, however, had detrimental affects on other native carnivores. It is well established that predators play a vital role in maintaining structure and stability of communities and that removal of predators can have a variety of cascading, indirect effects (Terborgh et al. 2001; Duffy 2003). Indeed, impacts of rangeland activities that have targeted predators for reduction to enhance livestock productivity extend far beyond the anticipated outcomes. Further, current investigations of trophic cascades resulting from the elimination of top predators can have implications beyond the immediate ecosystems occupied by those carnivores (Berger et al. 2001). Moreover, reduction of top carnivores can lead to unanticipated detrimental impacts to species that may otherwise not have been preyed on as a result of mesocarnivore release, whereby midsized carnivores benefit from a reduction in the numbers or densities of top carnivores (Berger et al. 2008). Thus, a consequence of the elimination of many carnivores from rangelands in North America has resulted in indirect impacts to other species and other than the rangeland ecosystems from which the carnivores in question were eliminated.

Rangeland Activities and Habitat for Native Ungulates

Because livestock and wild ungulates share rangelands, managers have examined the influence of cattle and domestic sheep on the vegetation used by white-tailed deer (*Odocoileus*

virginianus Zimmerman 1780), mule deer, elk, bighorn sheep (*Ovis canadensis* Shaw 1804), and pronghorn. In general, livestock using ranges shared with wildlife have historically had more negative than positive influences on ungulates, and grazing is not always considered an important conservation practice with beneficial outcomes. However, some studies examined how livestock influenced vegetation but did not present data related to how those influences altered productivity and recruitment of ungulates. Below are examples of studies that examined the use of prescribed grazing as a conservation practice for several ungulate species.

Pronghorn. Pronghorn populations have declined on the Anderson Mesa, Arizona, and cattle were considered a key factor in altering habitat. Five years after cattle were removed from Anderson Mesa, hiding cover (for fawns) increased by 8% at a distance of 5 m, but no differences were reported at 10 or 25 m (Loeser et al. 2005). Forb richness decreased in the fifth year after cattle removal by 16% but not in the following year, and canopy cover was unaffected. It will likely take longer than 5 yr of cattle absence to reverse damage that has occurred to this fragile environment, or some mechanism other than grazing was involved. However, pastures grazed by livestock conservatively or moderately were not used by pronghorn in New Mexico (Jamus et al. 2003).

In the Desert Experimental Range, Utah, pronghorn distribution was related to domestic sheep grazing, black sagebrush (*Artemisia nova* Beetle and Young), and topographic characteristics. Pronghorn-selected areas ungrazed by cattle and areas used moderately by sheep during dormant periods were not favorable for pronghorn (Clary and Beale 1983). Nevertheless, Mosley (1994) suggests that grazing rangelands by domestic sheep can be beneficial to wildlife habitat. However, Schwartz et al. (1977) suggest that pronghorn coexist on rangelands more successfully with cattle than with sheep.

White-Tailed Deer. Most of the studies examining livestock interactions with white-tailed deer have documented how deer respond to livestock under different grazing systems. From these data, conservation practices have been recommended. In general, white-tailed deer avoid livestock, and livestock operations are more profitable when deer are not considered in the operation (Bernardo et al. 1994). Conversely, returns from livestock were maximized when wildlife was not considered; however, small reductions in net gains (from livestock) can improve wildlife habitat (Bernardo et al. 1994).

The diets of white-tailed deer and cattle are different (i.e., deer consume forbs, and cattle consume more grass), and deer are more sensitive to grazing treatments than cattle. To enhance forage for white-tailed, cattle should be stocked at moderate rates with continuous grazing (or even less intensive grazing) to create environments where deer can select more forbs (Ortega et al. 1997a, 1997b). Dietary protein for growth and lactation of white-tailed deer was not met with short-duration or continuous grazing. However, the latter system may provide deer with more diversity and greater nutrition (Ortega et al. 1997b). Deer avoided concentrations of cattle and travel farther under short-duration than continuous grazing systems (Cohen et al. 1989). However, home ranges of white-tailed deer were not significantly different under short-duration or continuous grazing systems (Kohl et al. 1987). They also avoid anthropogenic water sources in short-duration grazing systems because of disturbance from humans, fences, and livestock (Kie 1991). Anthropogenic water sources for white-tailed deer should be on the periphery of short-duration grazing systems if it needs to be supplied (Prasad and Guthery 1986; Kie et al. 1991).

There are fewer studies examining how prescribed grazing by domestic sheep influenced white-tailed deer (Ekblad et al. 1993). In Texas, Darr and Kelebenow (1975) reported a negative relationship between domestic sheep and white-tailed deer due to removal of cover by the former.

Mule Deer. Overall, the best practice related to grazing for mule deer is to minimize cattle numbers on deer ranges. Moderate to heavy use of deer ranges by cattle reduced hiding cover (Loft et al. 1987), caused shifts in habitat (Loft et al. 1991, 1993), increased competition for forage (especially at high stocking rates and in

“livestock using ranges shared with wildlife have historically had more negative than positive influences on ungulates, and grazing is not always considered an important conservation practice with beneficial outcomes.”

Elk (*Cervus canadensis*) in Rocky Mountain National Park, CO. (Photo: David Briske)

dry years; Smith 1949; Kie et al. 1991; Yeo et al. 1993), and influenced foraging behavior (Loft et al. 1993; Kie 1996). Mule deer avoided pastures occupied by cattle (Wallace and Krausman 1987; Austin et al. 1983; Austin and Urness 1986; Bailey and Rogotzkie 1991).

However, several investigators examined how forage removal influenced mule deer and reported that mowing at 50% removal can increase grass and total biomass the following spring but that fall cattle grazing leaves more nutritious plants available in summer (Taylor et al. 2004). According to some, spring and summer deer ranges can be grazed by cattle an average of 70% (relative utilization) to enhance the ranges the following year (Short and Knight 2003). Burning can also enhance mule deer habitat (Williams et al. 1980).

Domestic sheep grazing deer ranges often benefit deer by improving forage quality in fall and increasing quantity in spring (Rhodes and Sharrow 1990). The degree of range improvement due to grazing by domestic sheep depends on the intensity of grazing and weather. Browse quality will improve with moderate grazing (40% to 55%) that ends by June (Alpe et al. 1999).

Guidelines to improve the quality of winter range for mule deer in the Great Basin were developed by Austin (2000) based on a review of grazing studies. The following guidelines were established to maintain or increase browse production on winter range.

1. Graze livestock between 1 May and 30 June.
2. Alternate grazing by class of livestock.
3. Use rest–rotation with yearly grazing 66% of the total rangeland.
4. Graze livestock to remove 50% of understory grasses and forbs.
5. Balance deer browsing in winter and livestock grazing in spring.
6. Monitor utilization using permanent plots.

Elk. Studies examining how livestock influence elk were similar to other ungulates examined; most work concentrated on the influence of livestock on forage and did not directly examine population effects. Overall, cattle use of elk ranges had little influence on forage quality when stocked at 3.7 ha · AUM^{-1}, but it did influence the quantity of forage available for elk (Dragt and Havstad 1987). Others (Wambolt et al. 1997) reported similar results when the nutritional values of forage were measured.

Understanding forage use by wildlife and livestock is important for wildlife and livestock management. Most studies of elk and cattle interactions examined use of pastures under different conditions. Because of the varied management plans for livestock, managers should address multiple herbivore species in relation to environmental and climatic variation (Werner and Urness 1998). For example, in Utah, elk did not influence available forage for cattle in June and August 1994, but use by cattle was greater in areas not used by elk in two of three rested pastures in June–August 1995. Cattle grazing reduced preferred winter elk forage in the initial growing season in Montana, but by the second season, the standing crop was similar to the ungrazed control (Jourdonnais and Bedunah 1990). Intensive cattle grazing in Washington decreased elk use of ranges in 1 of 3 yr by 28% (Skovlin et al. 1983).

Limited research has demonstrated how livestock grazing can improve elk forage and increase elk numbers. The Bridge Creek Wildlife Management area in northwestern Oregon was grazed by cattle without a

prescribed grazing system and supported 120 elk during winter over 13 yr. When a livestock grazing plan was initiated that incorporated rotational grazing, water distribution, properly located fences, salt placement, creation of a wildlife sanctuary, and closing roads, forage quality improved for elk and cattle, the elk population increased to nearly 1 200 animals, and AUM months for cattle grazing increased by 2.6 times (Anderson and Scherzinger 1975).

In other studies, elk shifted habitats when cattle were introduced (Wallace and Krausman 1987) and selected rested pastures over those used by cattle temporally (Frisina 1986; Yeo et al. 1993), even though fall cattle grazing and mowing (70% and 50% removal, respectively) can increase green vegetation the following spring (Frisina 1986; Short and Knight 2003; Taylor et al. 2004).

Impacts to elk range from domestic sheep depend on climatic conditions and grazing intensity. The quality of browse may improve with moderate grazing of sheep (40% to 55%) that ends by June (Alpe et al. 1999). Others (Rhodes and Sharrow 1990) suggest that at a stocking rate of 125 to 143 female-days · ha^{-1}, domestic sheep can improve forage quality in fall and forage quantity in spring. Carefully managed late-spring sheep grazing can improve winter forage quality on elk winter range (Clark et al. 2000).

Bighorn Sheep. Ranges used by bighorn sheep and cattle usually do not overlap spatially, but interactions have been documented (Halloran and Blanchard 1950; King and Workman 1984; Dodd and Brady 1986; Steinkamp 1990). Early reports (Halloran and Blanchard 1950) simply documented the occurrence of both animals, but later reports evaluated the relationships between them. Earlier studies of cattle and bighorn sheep (Spencer 1943; Halloran 1949; Matthews 1960; Arellano 1961) did not demonstrate competition. Habitat preferences for steeper slopes by bighorn sheep and gentler slopes by cattle precluded competition because there was no range overlap. However, Barmore (1962) argued that cattle grazing on gentle slopes has precluded the use of those areas by bighorn sheep, and Bleich et al. (1997) cautioned that extensive use of such areas could affect forage availability for male bighorn sheep in particular. Blood (1961) examined competition between cattle and bighorn sheep in Canada, where 70% of bighorn sheep winter range was used by cattle. He concluded that cattle grazing prevented increases in the bighorn sheep population.

King and Workman (1984) reported different associations between cattle and bighorn sheep in southeastern Utah. They reported bighorn sheep in higher, steeper, and more rugged talus slopes than cattle, which selected lower, gentler slopes and valleys close to roads and developed water sources. In addition, diets of the ungulates were different; cattle diets were dominated by grass, but bighorn sheep were browsers. King and Workman (1984) did not demonstrate that cattle and bighorn sheep competed for space or resources; however, they argued that the spatial separation they observed may result from a "social intolerance—avoidance factor." McCann (1956), Barmore (1962), McCullough and Schneegas (1966), Follows (1969), Ferrier and Bradley (1970), Dean (1975), Wilson (1975), Gallizioli (1977), and Albrechtsen and Reese (1979) argued that bighorn sheep avoid areas used by cattle. Steinkamp (1990) demonstrated that a translocated population of bighorn sheep clearly avoided cattle. As cattle moved into core areas used by bighorn sheep, sheep moved away. Additionally, the closer cattle grazed to sheep, the closer sheep remained near escape cover.

Social intolerance (Geist 1971) can have serious implications because cattle now graze most rangelands that historically supported bighorn sheep (Mackie 1978); 70% of public lands in the 11 most contiguous western states are grazed at least seasonally (US Department of the Interior 1986). Livestock grazing, even seasonally, appears to result in habitat fragmentation (Temple 1984), resulting in the exclusion of sheep from what is otherwise acceptable habitat. Bissonette and Steinkamp (1996) demonstrated that social intolerance can be a potent force influencing habitat use by sheep. Steinkamp's (1990) and Bissonette and Steinkamp's (1996) results pertain, however, to groups newly translocated into unoccupied habitat. Whether social intolerance between cattle and bighorn sheep is universal

remains equivocal. Resolution of the dispute is clouded by the almost universal disregard for spatial scale. Lack of consideration of scale effects can have profound implications for management. For example, in 1988–1989, the bighorn sheep population in Aravaipa Canyon, Arizona, was reduced by 52%. Mouton et al. (1991) examined the causes of mortality and concluded they were "probably the result of livestock related viral diseases compounded by nutritional stress." Because range overlap has been documented to result in sheep mortality by disease transmission, determination of overlap and the scale at which it occurs is most important. Overlap at the level of home ranges may have very different consequences from overlap on specific slopes or valley floor areas. Additionally, temporal overlap at different scales (e.g., seasonal and annual) would appear to have important ramifications for management.

In other areas of the Southwest, grazing by cattle has damaged bighorn sheep habitats (Gordon 1957; McColm 1963; Riegelhuth 1965; Gallizioli 1977). Low precipitation levels ensure that recovery of ranges will take many years, and in some areas damage from livestock grazing may be irreversible. Grazing by cattle has also influenced bighorn sheep habitat in less arid areas (Buechner 1960; Crump 1971; Geist 1971; Brown 1974) by converting grasslands to shrublands (Demarchi 1970).

Bighorn sheep do not do well when they share ranges with cattle. Following the population declines of bighorn sheep of the late 1800s and early 1900s, they did not recover as well as other native ungulates (e.g., mule deer). Bighorn sheep are not as tolerant as other native North American ungulates to poor range conditions, intraspecific competition, overhunting, and habitat alteration. In addition they are much more susceptible to diseases of livestock than other rangeland wildlife, especially diseases of domestic sheep.

Diseases of cattle that influence bighorn sheep are poorly documented, but diseases contacted from domestic sheep have played an important role in bighorn sheep mortality. Throughout the western United States, die-offs of bighorn sheep and population declines have occurred following the introduction of domestic sheep. Mortality was the result of competition for forage and space and shared diseases (Goodson 1982). According to Goodson (1982), "Co-use of ranges by domestic and bighorn sheep has been consistently linked with declines, dieoffs, and extinctions of bighorn populations from historic to recent times. While much of the evidence for competition between domestic sheep and bighorn sheep is circumstantial, it is sufficiently strong to have prompted management decisions against co-use of ranges by bighorn and domestic sheep by federal land management agencies and state wildlife departments." The Technical Staff of the Desert Bighorn Council (1990) reviewed 24 interactions between bighorn sheep and domestic sheep and found that bighorn sheep died as a result of all interactions. Recent experimental studies confirmed field observations; when bighorn sheep are exposed to domestic sheep, bighorns die from *Pasteurella haemolytica* (Foreyt 1989, 1990, 1992; Silflow et al. 1993; Foreyt et al. 1994).

The actual mechanisms that kill bighorn sheep after they come in contact with domestic sheep are poorly documented (Jessup 1985), but two trends appear clear (Technical Staff of the Desert Bighorn Council 1990): 1) a large portion of the bighorn sheep population dies, and (2) domestic sheep do not suffer ill effects because of their contact with bighorn sheep. Bighorn sheep are more susceptible to diseases they share with livestock. Domestic animals have been selectively bred for disease resistance, but bighorn sheep have not evolved with resistance to the complement of diseases they are now exposed in the presence of domestic stock. As a result, they have not developed effective immunity against livestock diseases. Silflow et al. (1991) examined domestic and bighorn sheep and concluded that they had different control mechanisms for lung metabolism, and differences in the metabolites released led to different regulation of lung defense mechanisms.

Disease. Biologists are not aware of all the factors creating negative interactions between domestic stock and bighorn sheep, but scabies, chronic frontal sinusitis, nematode parasites, pneumophilic bacteria, foot rot, parainfluenza III, bluetongue, sore mouth, paratuberculosis, and pinkeye are documented decimating factors to bighorn sheep (Jessup 1985).

Bighorn sheep have coexisted with humans for ≥ 30 000 years but now face a precarious future. They are an ecologically fragile species, adapted to habitats that are increasingly fragmented. Fragmentation of habitats increases when cattle share the same rangelands as bighorn sheep. Domestic sheep pose an even greater threat to bighorn sheep.

Small Mammals, Reptiles, and Amphibians

There are few studies that address specific conservation practices for small mammals, reptiles, and amphibians (i.e., access control, fences, closing mine shafts, and ponds). While limiting human access has positive effects on big game survival (Rowland et al. 2000), direct data are lacking for effects on small mammals, reptiles, and amphibians that are not directly harvested by humans. There is evidence that trampling caused by high amounts of human access (e.g., hiking and off-road vehicles) does affect the occurrence of small mammal species in montane (Liddle 1975) and urban habitats (Dickman and Doncaster 1987). Off-road vehicle use has been directly attributed to desert tortoise (*Gopherus agassizi* Cooper 1863) and Couch's spadefoot (*Scaphiopus couchi* Baird 1854) declines in California (Berry 1986). Such data suggest that controlling human use by limiting access may be effective in enhancing habitats for small mammals, reptiles, and amphibians.

In rangelands, fences often provide added vegetative cover resulting from different microclimates and seed deposition by birds (Holthuijzen and Sharik 1985). There is some research examining the effects of fences on small mammals. Merriam and Lanoue (1990) used radiotelemetry and showed that white-footed mice (*Peromyscus leucopus* Rafinesque 1818) in farmlands preferentially traveled along fencelines, even when associated vegetative structure was less than 1 m wide.

A primary concern should be for bat maternity roosts and hibernacula. Mohr (1972) provided data on the importance of these cave resources for bats, and Jagnow (1998) reviewed an example of effective closure to restrict human access but to leave openings for ingress and egress by bats.

Decreased vegetative cover at the edge of stock ponds resulting from cattle grazing was correlated with decreased abundance of Columbia spotted frogs (*Rana luteiventris* Thompson 1913) in Oregon (Bull and Hayes 2000). Livestock effects on water quality were correlated with decreased larval diversity and abundance of amphibians in Tennessee (Schmutzer et al. 2008). However, the effect of cattle on terrestrial habitat quality and postmetamorphic survival of amphibians is yet to be quantified.

Prescribed Grazing and Upland Wildlife Habitat Management

Small Mammals. The greatest proportion of literature documenting effects of grazing on small mammals has focused on rangeland and riparian areas in the western United States. The density of aboveground biomass is important in structuring small mammal communities. Grant et al. (1982) indicated that small mammal communities and their response to grazing varied widely. Tallgrass communities tend to occur in areas of reliably high soil moisture and provide a high ratio of vegetation to seed with large accumulations of litter. These communities support highly variable populations of herbivorous litter-dwelling small mammals with high reproductive rates that can consume large amounts of vegetation. Grasslands of intermediate productivity have low biomass and low diversity of omnivorous and primarily surface-living small mammals, but both forage consumption and reproductive output are somewhat lower than in tallgrass prairie. Shortgrass prairie supports high biomass and high diversity of relatively long-lived omnivorous or granivorous species that reproduce opportunistically with precipitation and that use available resources (seeds and insects) intensively. Communities that differ in species composition, niches, and trophic dynamics are expected to differ in their responses to grazing. Land managers should anticipate that small mammals associated with herbaceous or shrub cover will decline when cattle remove this cover (Moulton et al. 1981; Giuliano and Homyack 2004; Johnston and Anthony 2008). Livestock grazing removes standing plant biomass but also prevents accumulation of ground litter that may influence small mammal community

Black-tailed prairie dogs (*Cynomys ludovicianus*) are common throughout the Great Plains and grazing facilitates colony expansion. (Photo: David Briske)

Rocky Mountain bighorn sheep (*Ovis canadensis*), Yellowstone National Park. (Photo: Jerod Merkle)

composition, plant species growth, and seedling establishment via shading and changes in soil temperature and moisture (Fowler 1988). In southwestern grasslands and shrublands, grazing and fire result in rodent communities dominated by heteromyids (family Heteromyidae; pocket mice, kangaroo rats, and kangaroo mice) instead of murids (family Muridae; rats, mice, hamsters, voles, lemmings, and gerbils) on mesic sites. In more arid sites, grazing and fire favor kangaroo rats (*Dipodomys* spp.) over pocket mice (*Perognathus* spp.; Jones et al. 2003).

Most studies demonstrating negative impacts on small mammal populations have attributed those effects to changes in vegetation cover and perceived predation risk (Grant et al. 1982; Uresk and Bjugstad 1983; Heske and Campbell 1991; Hayward et al. 1997) or to long-term changes in plant species diversity (Jones and Longland 1999). However, Steen et al. (2005) provided evidence that forage competition occurs between livestock and voles, herbivores of greatly differing size. Grazing can either increase or decrease plant community heterogeneity (Adler et al. 2001). Detling (2006) provided the most extensive review of our state of knowledge concerning livestock and prairie dog interactions and concluded that we still cannot accurately determine the effect of prairie dogs on domestic livestock production. However, there is evidence that heavy livestock grazing can facilitate prairie dog colony expansion. Lomolino and Smith (2004) reported that prairie dog colonies had similar species richness of nonvolant mammals, reptiles, and amphibians as adjacent landscapes in Oklahoma but harbored different and more rare and imperiled species. Milchunas et al. (1998) suggested that livestock grazing impacts on other grassland herbivores may depend, in part, on temporally variable short-term trade-offs between plant quantity and plant nutrient quality. Habitat productivity and herbivore densities may mediate shifts from facilitative to competitive interactions between different-sized herbivores (Krueger 1986; Cheng and Ritchie 2006). Field voles (*Microtus agrestis* L. 1761) in Denmark showed a skewed quadratic response to grazing intensity (Schmidt et al. 2005) with population biomass and productivity at light to intermediate grazing intensity slightly greater than ungrazed and much greater than heavily grazed sites. Grazing

on these sites reduced thick vegetative cover and promoted more nutritional regrowth, and this species of vole responded much the way livestock do. Steen et al. (2005) reported that field voles in Norway responded similarly but that bank voles (*Clethrionomys glareolus* Schreber 1780), whose diet differs, did not respond to sheep grazing.

Reptiles and Amphibians. Kazmaier et al. (2001) detected no differences in survival or demography of Texas tortoises (*Gopherus berlandieri* Agassoz 1857) between moderately grazed (short-duration, winter–spring rotational grazing regime; 6–28 AUM $d \cdot ha^{-1} \cdot yr^{-1}$) and ungrazed sites in the Western Rio Grande Plains, Texas. Brodie (2001) examined freshwater turtles across North America and suggested that increased siltation and soil compaction resulting from overgrazing in riparian areas could impact reproduction of freshwater turtles.

Smith and Ballinger's (2001) review indicated that lizards that sit and wait in open habitats (e.g., collared lizard [*Crotaphytus collaris* Say 1823], lesser earless lizard [*Holbrookia maculate* Girad 1851], and side-blotched lizard [*Uta stansburiana* Baird and Girard 1852]) tend to be positively affected at the population level by livestock grazing, whereas active foragers that need vegetative cover (e.g., western whiptail [*Cnemidophorus tigris* Baird and Girard 1852], western stone gecko [*Diplodactylus granariensis* Starr 1879], fine faced gecko [*Diplodactylus pulcher* Steindachner 1870], desert spiny lizard [*Sceloporus magister* Hallowell 1854], bunch grass lizard [*Sceloporus scalaris* Weigmann 1828], and Baja California bush lizard [*Urosaurus nigricaudus* Cops 1854]) tend to be negatively affected. Fair and Henke (1997) indicated that Texas horned lizards (*Phrynosoma cornutum* Harlan 1825) selected for burned plots and did not select for grazed plots in southern Texas. Lizard community composition in Arizona and desertified arid grasslands (Castellano and Valone 2006) was significantly different between inside and outside a grazing exclosure. Analysis of tail-break frequencies suggested that higher predation rates outside the exclosure may have contributed to increased abundance of eastern fence lizard (*Sceloporus undulutes* Bosc and Daudin 1801) and side-blotched lizards following livestock removal. In contrast, the round-tailed horned lizard (*Phrynosoma modestum* Girard 1852) was significantly less abundant inside the exclosure.

Knutson et al. (2004) reported that small agricultural ponds in southeastern Minnesota provided breeding habitat for at least 10 species of amphibians. Gray et al. (2004) reported that relative abundance (i.e., average daily capture) of New Mexico and plains spadefoot toads (*Spea multiplicata* Cope 1863 and *S. bombifrons* Cope 1863) was greater at cropland than at grassland playas but that the abundance of other species and diversity of the amphibian assemblage was not affected by surrounding land use. However, Gray and Smith (2005) reported that mass and length of amphibians from playas surrounded by grasslands were greater than those from agricultural playas. They attributed this to altered hydroperiod in playas surrounded by agriculture. Body size is positively related to the probability of survival, reproduction, and evolutionary fitness in amphibians (Gray et al. 2004). Thus, if cultivation of landscapes surrounding wetlands negatively influences postmetamorphic body size of amphibians, restoration of native grasslands surrounding playa wetlands may help prevent local amphibian declines.

Restoration and Management of Rare or Declining Habitats

Manipulating riparian herbaceous cover and stream habitats are conservation practices that have influenced small mammals, reptiles, and amphibians. Endangered Columbia Basin pygmy rabbits (*Brachylagus idahoensis* Merriam 1891) avoided grazed areas with fewer burrows than ungrazed areas (Thines et al. 2004). Grazing and mowing have been used effectively in specific cases to improve habitat for small mammal and reptile species that prefer reduced vegetative cover. Grazing reduced herbaceous and woody cover for the endangered Stephen's kangaroo rat (*Dipodomys stephensi* Merriam 1907) in California (Kelt et al. 2005) and reduced rhizomatous plant growth to facilitate burrowing while increasing sunning spots for threatened bog turtles (*Clemmys muhlenbergii* Schoepff 1801) in New Jersey (Tesauro 2007).

Riparian Herbaceous Cover. Medin and Clary (1989) reported that, after 11 yr of grazing exclusion, small mammal biomass

“there are common research needs and recommendations that need to be considered if administrators, managers, biologists, and the public are to better understand how the conservation practices of NRCS apply to upland wildlife on western rangelands in the United States”

was three times greater on an ungrazed aspen (*Populus tremuloides* Mrchx.) and willow (*Salix* spp.) riparian site. Chapman and Ribic (2002) reported that ungrazed stream bank buffer strips supported more small mammals and species than similar grazed areas and that rotational grazing was not different from continuous grazing as applied to small mammal responses. Grazing in wet meadows can have indirect effects on small mammals also. Whitaker et al. (1983) reported that ground-dwelling and fossorial invertebrates in diets of vagrant shrews (*Sorex vagrans* Baird 1857) were replaced primarily by volant species on grazed sites. Klaus et al. (1999) reported that grazing in Wyoming and Montana did not affect reproductive activity but did affect survival of young water voles (*Microtus richardsonii* DeKay 1842), a species of management concern in alpine riparian habitats. Land managers should anticipate that small mammals associated with herbaceous or shrub cover in riparian areas will decline when cattle remove this cover (Moulton et al. 1981; Giuliano and Homyack 2004; Johnston and Anthony 2008). Frog community response to grazing intensity was positively correlated with grazing reduction of palustrine vegetation in an Australian floodplain (Jansen and Healey 2003).

Stream Habitat Improvement and Management. Homyack and Giuliano (2002) reported that northern queen snakes (*Regina septemvittata* Say 1825) and eastern garter snakes (*Thamnophis sirtalis* L. 1758) were more abundant on fenced than unfenced stream banks but that most herptofauna may require longer than their 4-yr study to respond to exclusion from grazing. As with other conservation practices, little is available on how they influence wildlife populations.

RECOMMENDATIONS

Little peer-reviewed research exists that examines the effects of conservation practices on habitat heterogeneity and diversity of wildlife. Most studies that we reviewed failed to collect pretreatment data, lacked experimental controls, had limited or no replication, or were too short in duration. Implications may often be extrapolated too broadly because results are frequently derived from studies of local management actions.

Research needs and recommendations for the different groups of fauna vary. However, there are common research needs and recommendations that apply to all categories that need to be considered if administrators, land use planners and managers, biologists, and the public are to better understand how the conservation practices of NRCS apply to upland wildlife on western rangelands in the United States.

1. Experimentally designed studies with replicates and controls are necessary. These studies need to be conducted so that scientifically reliable data can be collected.
2. Studies have not been designed to understand how NRCS conservation practices apply to wildlife. This can be acquired only through targeted research. Specific studies should be designed to determine how specific NRCS conservation practices influence wildlife and the habitat they depend on, including (but not limited to) access control, access road, brush management, clearing and snagging, conservation cover, diversion, early successional habitat development/management, fence, hedgerow planting, herbaceous weed control, land clearing, reclamation, mine shaft and adit closing, pond, range planting, restoration activities, spring development, tree and shrub establishment, and upland wildlife habitat management.
3. Carnivore management is not an aspect of NRCS conservation practices for upland wildlife, but because of their role in the ecosystem, they need to be considered and managed.
4. One common theme that is constantly emphasized in management theory is the importance of monitoring. Unfortunately, funds are not provided for these important activities. As a result, projects and practices are put in place, management plans are developed, and short-term research is conducted with little or no follow-up. This lack of efficient monitoring creates numerous information gaps that otherwise may have been filled. It is critical that monitoring be included in local, regional, and national management efforts so that the results of those efforts can be determined.

KNOWLEDGE GAPS

Game Birds

1. Experimental evidence of grazing practices beneficial to game birds is largely lacking. Before–after control–impact field experiments are needed to determine widespread, relative effects of grazing treatments and stocking intensities on nesting success and female and chick survival (Beck et al. 2000). Investigations also are needed to evaluate effects of grazing, use levels, and stocking rates on abundances of important forbs and insects in brood-rearing habitat because these responses are poorly understood. Experiments should be well replicated and of a sufficient time to understand short- and long-term effects on populations.
2. Similarly, investigations are needed to understand how to reduce and mitigate impacts of energy development and other significant sources of human disturbance over large landscapes as they relate to conservation practices. Recent studies show the large-scale and population-level impacts of oil and gas development on wildlife, including mule deer (Sawyer et al. 2009), sage-grouse (e.g., Walker et al. 2007), and songbirds (Ingelfinger and Anderson 2004). Wind energy will reduce our carbon footprint, but impacts to wildlife resulting from roads, noise, tall turbines, and additional power lines are poorly understood (e.g., lesser prairie-chicken; Pruett et al. 2009). These studies also will require strong statistical designs that include treatments and controls at spatial and temporal scales relevant to landscape-scale impacts (Johnson and St-Laurent 2011).
3. A multitude of local-scale questions should be addressed as part of larger investigations. For example, we should determine whether the addition of anthropogenic water sources benefits quail (and other wildlife) populations in the desert Southwest (Western Quail Management Plan 2008) and whether mortality from fence collisions places a role in population dynamics, and, if so, we should develop recommendations on type and placement of fencing to reduce mortality (Wolfe et al. 2007). Studies should be conducted long enough to capture the short- and long-term influences that impact the practice being examined.

Researchers should collaborate with management agencies to develop large and experimental projects as part of treatment projects planned by state and federal partners. In response, researchers and agencies can commit to monitoring at appropriate scales to evaluate treatment effects and to provide a basis for adaptive management.

Nongame Birds

As noted above, few experimental studies have specifically evaluated the use of rangeland management to benefit nongame birds. Most efforts have been from the midwestern United States in the series of studies conducted by Herkert et al. (1996, 2003). The degree to which their results apply to western ecosystems is unknown. To date, most studies conducted in the West have consisted of "fence-line" observational studies whereby investigators compare adjoining pastures with and without cattle grazing. Questions concerning grazing regime, timing (both longevity and season of grazing), stocking levels, and related variables have yet to be addressed. To do so will require well-designed, replicated studies that can determine various sources of variation to understand cause–effect relationships.

Bobcats (*Lynx rufus*) are a common rangeland predator that subsist primarily on rodents, rabbits, and birds. (Photo: Tim Fulbright)

Carnivores

Fruitful areas of research will include further evaluations of the role that top carnivores play in ecosystem structure and function (Hebbelwhite et al. 2005) and understanding the benefits or consequences of restoring those predators to historically occupied distributions. Additionally, better understanding of conditions that result in conflicts between humans and large carnivores (Wilson et al. 2006) may provide opportunities to lessen conflicts in the future. Continued efforts to improve methods of reducing human–carnivore impacts and the implementation of those methodologies on rangelands is desirable and necessary to conserve large carnivores (Shivik 2006). Further, responses of small carnivores to conservation practices should be explored more explicitly because of their importance as predators of ground-nesting birds; currently, much of the literature addresses risk of predation to avian species associated with rangeland management practices rather than demographic or habitat shifts in small carnivores that may result in those shifts in predation risk.

Ungulates

Much of the peer-reviewed literature documents the influence of livestock and wildlife on range flora, but the studies are usually not replicated, are conducted on a small scale, and do not indicate how associations with livestock influence productivity and recruitment of wildlife.

Additional research is needed to address these issues. In addition, because of the fragmentation of bighorn sheep habitat by livestock (Steinkamp 1990; Bissonette and Steinkamp 1996), social intolerance (Geist 1971), and disease transmission (Jessup 1985), most researchers argue that prescribed livestock grazing should not occur in bighorn sheep habitat. To minimize avoidance of livestock by bighorn sheep and, hence, avoidance of habitat, livestock and bighorn sheep should not be close to each other (Steinkamp 1990; Bissonette and Steinkamp 1996). When separation is not possible, efforts should be made to minimize contact (e.g., placement of anthropogenic water sources or fencing critical areas), monitor distribution, monitor range conditions, and carefully watch for incidences of disease outbreaks (Goodson 1982; McCullough et al. 1980; Technical Staff of the Desert Bighorn Sheep Council 1990).

Small Mammals, Reptiles, and Amphibians

Responses of small mammals, reptiles, and amphibians to grazing and other range management practices is species and often species-habitat specific. Few general trends have been identified, as studies have not been adequately designed to understand the underlying processes responsible because of the highly variable population dynamics of these groups of organisms and poor experimental designs (Johnson 1982). Experiments need to be of sufficient duration (perhaps on the order of decades in some ecosystems) and sufficient replication (over broad regional ranges) to isolate effects of interacting environmental factors that are usually not subject to experimental control from the effects of rangeland "treatments" (Rosenstock 1996). At least four avenues would assist in better data:

1. Experimental evidence of conservation practices beneficial to small mammals, reptiles, and amphibians is largely lacking. Experiments designed with pre- and posttreatment data and controls are needed to determine relative effects of treatments on abundance and reproductive success of wildlife species. Experiments must include regional replications and be of sufficient duration to account for the variable nature of small animal populations to enable managers to understand short- and long-term population effects attributable to conservation practices at regional levels.
2. Monitoring the distribution of various land uses in different landscapes (e.g., clumped or dispersed) and at what scale they occur are crucial for assessing long-term population persistence of small mammals, reptiles, and amphibians in fragmented landscapes.
3. When examining the effects of a management practice, comprehensive analyses, including the impacts of type, frequency, timing, and extent of disturbances (e.g., mowing, burning, or grazing) of vegetation, are necessary to understand the species and species-site-specific effects of such practices on species

abundance and reproductive success.

4. Researchers should collaborate with management agencies to develop large-scale, cost-effective experimental projects in an adaptive resource management strategy as part of conservation projects planned by state and federal partners. Commitments need to be made for monitoring at appropriate scales to evaluate treatment effects and to provide sound scientific data of sufficient scope and scale for assessing the true effects of conservation practices.

CONCLUSIONS

Very few of the 167 conservation practices listed by the NRCS have been evaluated in the peer-reviewed literature to determine their influence on upland wildlife. Activities associated with those conservation practices, particularly those efforts to enhance livestock production by limiting predation, have not been adequately investigated with respect to their overall impacts to rangeland ecosystems. Nevertheless, rangelands are important for protecting biodiversity, suggesting that future conservation efforts may require less reliance on reserves and a greater focus on private lands (Maestas et al. 2003). Grazing by livestock has received more attention in the literature than other conservation practices, but even then, studies often fail to distinguish between the different types, seasons, and intensities of grazing. Peer-reviewed literature evaluating how conservation practices influence upland wildlife habitat management has not received high priority, and their complex influences on wildlife and its habitat are largely unknown. Furthermore, other uses of rangelands (e.g., energy development) result in broad-scale loss and degradation of habitat that overwhelms other types of management (e.g., conservation practices) by increasing predation rates, promoting the spread of invasive plants, and facilitating disease transmission. However, the use of rangelands for sustainable livestock production has the potential to ensure the maintenance of wildlife habitat, especially when compared to energy development and urbanization, which will ensure that wildlife habitat will persist into the future.

Studies will need to be designed as targeted research, with adequate replicates and controls, for outcome-based science if managers and scientists are to better understand how NRCS conservation practices influence wildlife on western rangelands. Future studies should also follow rigorous before–after control–impact designs, be implemented at the landscape level, and be conducted for a sufficient amount of time to understand how NRCS conservation practices influence ecosystem dynamics.

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CHAPTER

7

Invasive Plant Management on Anticipated Conservation Benefits: A Scientific Assessment

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Invasive Plant Management on Anticipated Conservation Benefits: A Scientific Assessment

7

Roger L. Sheley, Jeremy J. James, Mathew J. Rinella, Dana Blumenthal, and Joseph M. DiTomaso

INTRODUCTION

Invasive plant species have many negative impacts on rangelands throughout the world. Invasive plants can displace desirable species, alter ecological processes, reduce wildlife habitat, degrade riparian systems, and decrease productivity (DiTomaso 2000; Masters and Sheley 2001). Invasive plants are estimated to infest about 100 million ha in the United States (National Invasive Species Council 2001). Experts recognize invasive species are the second most important threat to biodiversity after habitat destruction (Pimm and Gilpin 1989; Randall 1996; Wittenberg and Cock 2001). Furthermore, Wilcove et al. (1998) estimate invasive species have contributed to the placement of 35% to 46% of the plants and animals on the federal endangered species list. In 1994, the impacts of invasive plant species in United States were estimated to be $13 billion per year (Westbrooks 1998). The amount of land infested by invasive plants is rapidly increasing (Westbrooks 1998) and subsequently the negative impacts of invasive plants are escalating. To address this issue, federal agencies and private land managers have developed and implemented integrated pest management (IPM) programs on rangeland.

Description of IPM on Rangeland

IPM is a long-standing, science-based, decision-making process that identifies and reduces risks from pests and pest management–related strategies (USDA Regional IPM Centers 2004). It was initially used to describe agricultural systems, but has since expanded to include wildlands and rangelands. IPM processes involve the coordinated use of pest biology, environmental information, and management technologies to prevent significant pest damage through economical means, while at the same time posing minimal risk to people, property, resources, and the environment.

In recent years, invasive plant management has evolved to more frequently incorporate an IPM philosophy, as opposed to focusing on a single control option with little consideration of the ecosystem or the side effects of particular control methods. Although IPM approaches are not currently used in many regions, research has shown that integrating various combinations of control options can provide more effective control compared to a single option.

IPM strategies have been used in rangelands for at least 20 yr, with interest in this approach greatly increasing over the last decade. Examples of more current IPM approaches include the combination of biological control agents (Lym and Nelson 2002; Nelson and Lym 2003; Wilson et al. 2004; DiTomaso 2008; Joshi 2008), prescribed burning (DiTomaso et al. 2006a), grazing (Sheley et al. 2004), mowing (Sheley et al. 2003; Renz and DiTomaso 2006), and revegetation (Enloe et al. 2005). Many of these integrated approaches combine nonchemical strategies with judicious use of herbicides. In most of these cases, the goal of the IPM approach was to establish a more desirable plant community that not only provides necessary ecosystem functions, but also provides some resistance to reinvasion, and thus, more effective long-term management of invasive plant species.

Description of Assumed Conservation Benefits of IPM on Rangeland

Specific plant species have been perceived as weeds since agriculture began about 10 000 yr ago. Early agriculturalists used hoes and grubbing implements to control weeds for

Invasive species have the capacity to create complete monocultures like the field of medusahead near Boise, Idaho pictured above. (Photo: Alex Boehm)

the specific benefit of increased commodity production (Radosevich et al. 1997). During the 20th century, synthetic organic chemicals were extensively used to control invasive weeds in rangeland production systems, and natural enemies of invasive weeds have been used to reduce some undesirable plant populations below an economic threshold. The primary objective was to enhance grass production, while minimizing adverse ecological and human impacts of the control effort.

Currently, ecologists and land managers recognize substantial adverse ecological, environmental, and economic impacts associated with invasive plants (Pimentel et al. 1999; DiTomaso 2000; Levine et al. 2003). In response, they have designed more-comprehensive invasive plant management approaches in an attempt to achieve an array of benefits in addition to enhanced control and grass production. Invasive weed prevention strategies and programs are aimed at protecting noninfested rangeland. A major focus has been to manage invasive plants to establish and/or maintain a desired plant community, especially to promote restoration of natural plant communities. The assumed benefit of restoring desired vegetative cover is to create and maintain healthy functioning ecosystems that reduce reinvasion, protect soils, control erosion, reduce sediment, improve water quality and quantity, and enhance stream flow. Invasive plant management also aims to benefit biological diversity and wildlife through habitat improvement.

OBJECTIVES AND APPROACH

The objective of this chapter is to provide a comprehensive assessment of the degree to which invasive plant management is achieving several commonly anticipated and desired benefits. We used a comprehensive review of peer-reviewed literature to assess the efficacy of various invasive plant management practices for each of nine conservation purposes developed for the Natural Resources Conservation Service (NRCS) conservation practice standard of herbaceous weed control. In contrast to the other conservation practice standards, this one was developed simultaneously with the Conservation Effects Assessment Program, so the stated purposes do not directly match those in the new standard. This new conservation practice standard is defined as the removal or control of herbaceous weeds including invasive, noxious, and prohibited plants. The writing team developed the following conservation purposes at the request of the NRCS for this chapter: 1) protect noninfested rangeland; 2) enhance quantity and quality of commodities; improve forage accessibility, quality and quantity for livestock; 3) control undesirable vegetation; 4) create a desired plant community; 5) change underlying causes of weed invasion; 6) restore desired vegetative cover to protect soils, control erosion, reduce sediment, improve water quality and quantity, and enhance stream flow; 7) maintain or enhance wildlife habitat including that associated with threatened and endangered species; 8) protect life and property from wildfire hazards; and 9) minimize negative impacts of pest control on soil resources, water resources, air resources, plant resources, and animal resources. The chapter also contains a section detailing recommendation and knowledge gaps, and conclusions addressing this conservation practice.

ASSESSMENT OF INTEGRATED PEST MANAGEMENT CONSERVATION PRACTICES

Protecting Noninfested Rangeland

Invasive plant management has traditionally focused on controlling invasive plants on already-infested rangelands, with less emphasis placed on protecting noninfested rangeland by preventing invasions (Zalvaleta 2000; Peterson and Vieglasis 2001; Simberloff 2003). A proactive approach focused on systematic prevention and early control provides solid economic returns where, on average, every dollar spent on early intervention prevented $17 in later expenses (OTA 1993). The major components of invasive plant prevention programs include minimizing invasive plant introduction into noninfested areas (often through vector management), early detection and eradication of satellite patches, and increasing the resistance of desirable plant communities and soil systems to invasion (Davies and Sheley 2007).

Minimizing Invasive Plant Introductions. A substantial amount of literature documents the

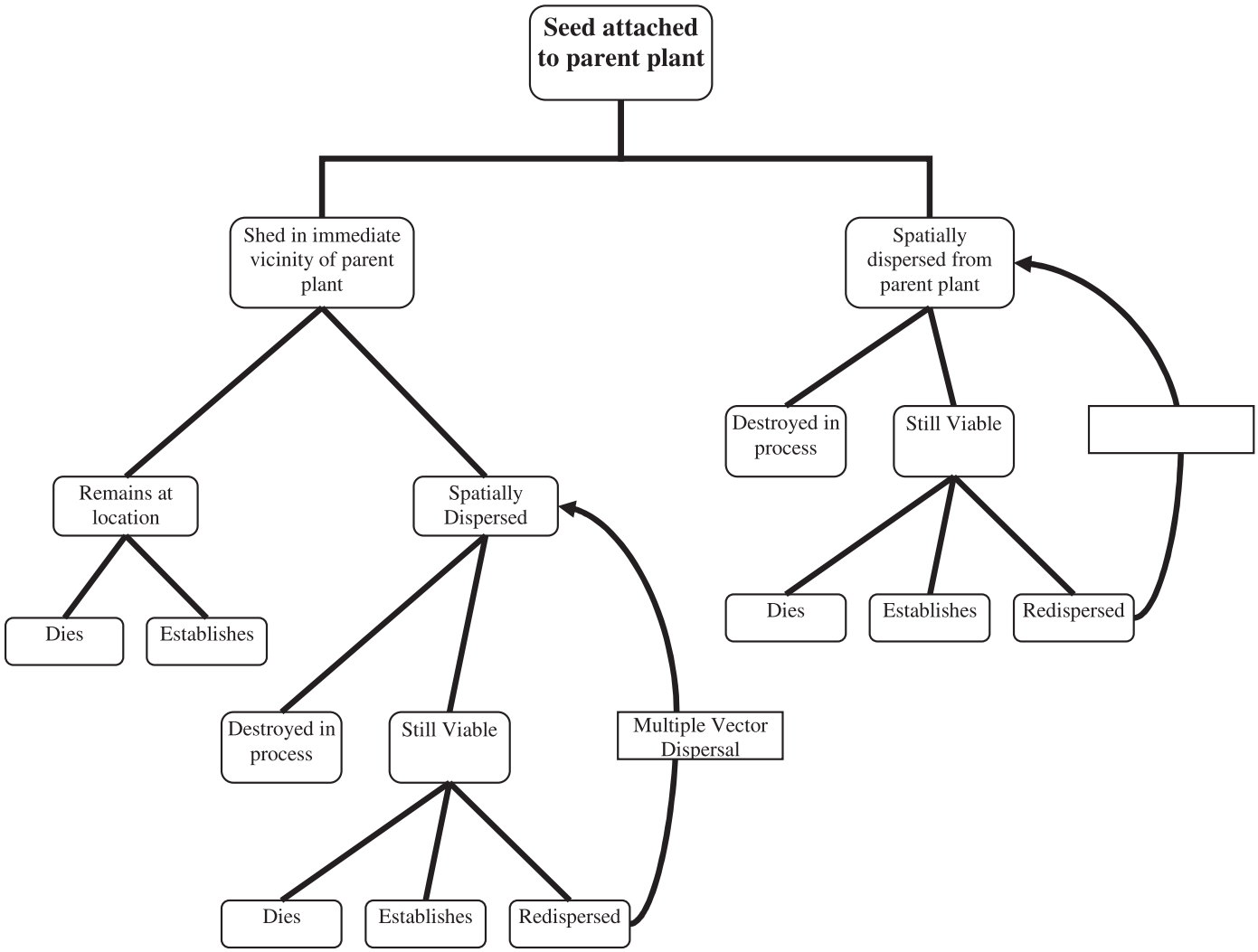


FIGURE 1. The potential fates and pathways of seed.

modes of plant dispersal throughout the world (Riley 1930; Janzen 1982). National, regional, and local introductions of invasive plants can occur many different ways (Plummer and Keever 1963). The most successful methods for reducing introductions of invasive plants are to create a break or diversion, especially in short- and long-distance dispersal (Fig. 1; Davies and Sheley 2007). Identifying vectors that are major dispersers of an invasive plant species provides vital information necessary for interrupting dispersal to new areas (Wittenberg and Cock 2001; Ruiz and Carlton 2003).

Dispersal vectors for some invasive species are known (Selleck et al. 1962; Brown and Archer 1987; Miller 1996; Kindschy 1998), but there is an obvious paucity of information about dispersal vectors. Regardless, Davies and Sheley (2007) provide a conceptual framework for preventing spatial dispersal of invasive plants. The framework identifies major potential vectors by incorporating invasive plant seed adaptations for dispersal through space and infestation locations relative to vector pathways (Fig. 2). Land managers can use the framework to guide efforts to limit dispersal of invasive plant seeds where it is possible.

A major weakness in invasive plant management is our lack of knowledge about the efficacy of various prevention strategies. Tests of methods of preventing dispersal are extremely rare in the literature; however, most studies identifying dispersal vectors intuitively suggest a method to minimize these vectors.

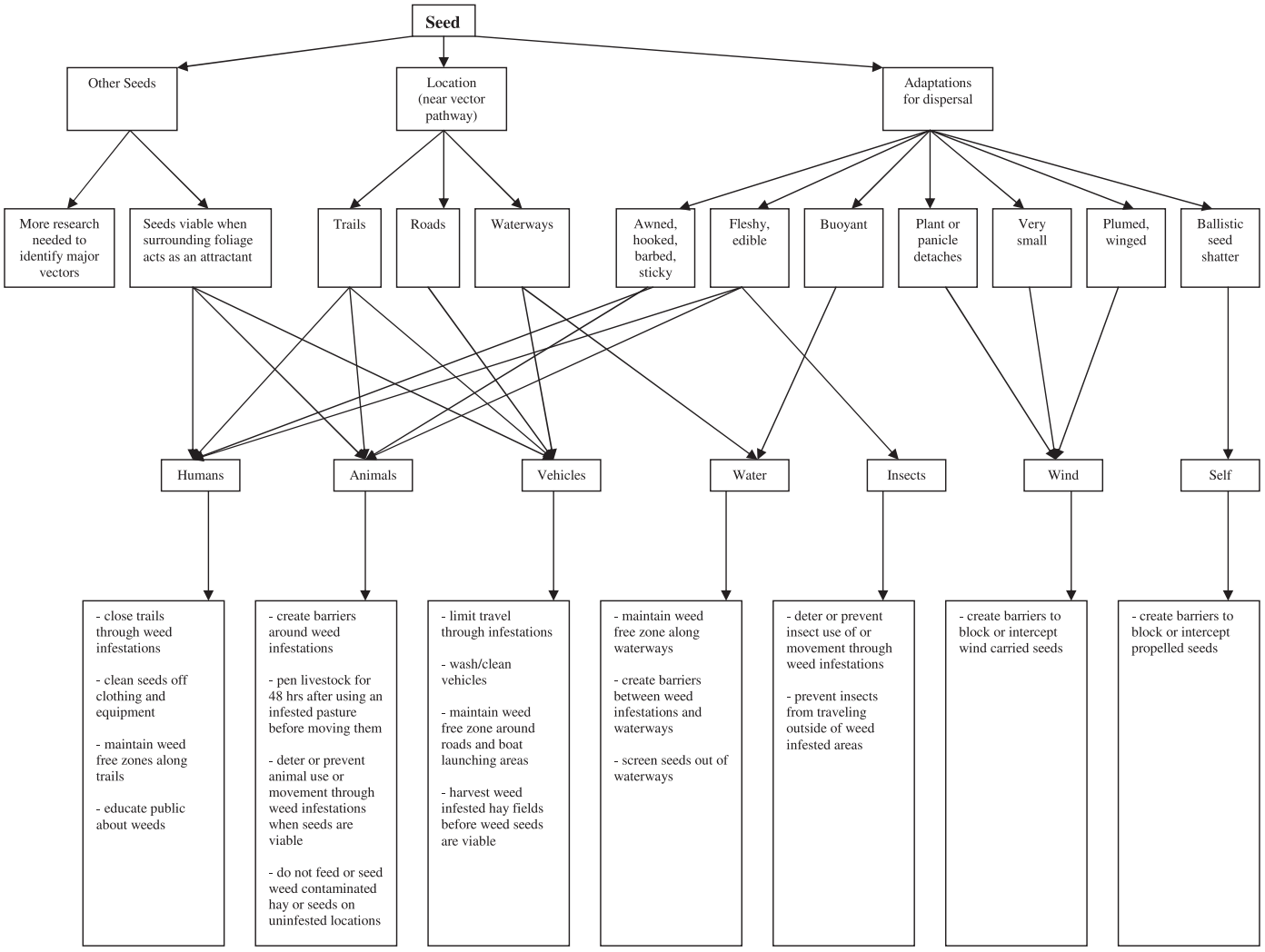


FIGURE 2. Conceptual framework that identifies the major dispersal vectors of invasive plant species and the corresponding dispersal management strategies.

For example, avoiding human or livestock contact with invasive species possessing hooks, barbs, and awns during seed production will likely help minimize dispersal (Agnew and Flux 1970; Sorensen 1986). We could only find a single study that directly tested a potential prevention strategy. In that study, wind dispersal was limited by increasing neighboring vegetation height for species having large plumes (Davies and Sheley 2007).

Early Detection and Rapid Control Response. One key to preventing new infestations is early detection of small patches that have a high probability of expanding into large infestations (Moody and Mack 1988). Early detection occurs at multiple levels of organization. The United States implements a national pest survey and detection program through the US Department of Agriculture–Animal and Plant Health Inspection Service. On the local level, early detection is difficult and requires educated and well-informed land managers, pest management specialists, and private land owners (Navaratnam and Catley 1986). Systematic weeds surveys (Johnson 1999), mapping based on sampling (Roberts et al. 2004), global positioning systems (Lass and Callihan 1993), and remote sensing (Steven 1993) have all been used to detect new infestations of invasive plants. In spite of the importance of detecting small infestations of invasive weeds, the cost, difficulty of implementation, and lack of reliable technology limit effective local early detection programs.

It is critical that small patches be effectively eradicated quickly after they have been located (Zamora et al. 1989; Simberloff 2003). Eradication involves the destruction of every

individual from an area (Newsome and Noble 1986). Most eradication strategies include aggressively repeated monitoring and control procedures (Weiss 1999a). There is a paucity of successful plant eradications found in the scientific literature and most descriptions of eradication programs are published in non–peer-reviewed formats (Simberloff 2003). Very few successful examples exist in the literature. The first international conference on eradication reported about 15 plant species that were eradicated from various areas around the world (Simberloff 2001). Pokorny and Krueger-Mangold (2007) provide evidence that small-scale eradication is achievable by documenting the successful removal of dyer's woad (*Isatis tinctoria* L.) from various counties in Montana. One biological tenet of successful eradication is that the infestation must be in the initial phases of invasion and only dominate a small area. Removing an invasive species is possible, especially for small infestations, but only under some circumstances and with potentially unpredictable results (Myers et al. 2000).

Invasion-Resistant Plant Communities and Soil Systems. Promoting desired species is a critical component of invasive plant management, especially in an attempt to prevent invasions (Sheley et al. 1996). Researchers have shown that functionally diverse plant assemblages resist invasion better than less-diverse assemblages (Burke and Grime 1996; Levine and D'Antonio 1999; Kennedy et al. 2002; Pokorny et al. 2005). Invasion-resistant plant communities can be achieved by maximizing niche complementarity among desired species (Tilman et al. 1997; Brown 1998; Carpinelli 2001; Fargione and Tilman 2005; Funk et al. 2008). Furthermore, those plant communities that maximize biomass production also minimize invasion (Hooper and Vitsousek 1997; Anderson and Inouye 2001).

Strategies aimed at maintaining desired plant communities help protect noninfested rangeland. For example, prescribed fall burning of late-seral big sagebrush–bunchgrass plant communities stimulated the herbaceous component and increased the resistance of the communities to cheatgrass invasion 4 yr postburn (Davies et al. 2008). In another example, clipping to simulate grazing greatly reduced medusahead (*Taeniatherum caput-medusae* L.) by removing decadent material for desired plant species, which stimulated regrowth and enhanced competitive ability (Sheley et al. 2008).

It has been proposed that invasion-resistant soils can be created by lowering plant-available nitrogen (Vasquez et al. 2008). Managing soil-available nitrogen can be achieved by light to moderate levels of grazing. Grazing animals can remove nitrogen in plant material, making it unavailable to plants (Neff et al. 2005; Steffens et al. 2008). Mowing can remove nitrogen if the plant material is removed from the site (Oomes 1990; Moog et al. 2002).

Conclusions and Management Implications. Protection of noninfested rangeland is central to the successful implementation of any integrated weed management program. Achieving the actual protection benefit is possible, but difficult, primarily because of a lack of effective techniques to interrupt dispersal vectors and our inability to detect new infestations before they become large infestations. Once found, small patches of invasive plants can be eradicated, but a comprehensive and intensive long-term eradication program must be employed. Many social, technological, and economic barriers exist that minimize the success of eradicating large infestations. The most scientifically developed strategy for protecting noninfested rangeland from invasion are those that convey some degree of invasion resistance to the plant community and possibly soils Managing desired plant communities to enhance the success of late-seral species, enhance diversity, and maximize productivity should help to minimize invasion and protect noninfested rangeland.

Enhance Quantity and Quality of Commodities; Improve Forage Accessibility, Quality, and Quantity for Livestock

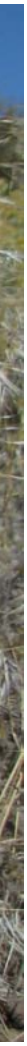
The primary marketable commodities garnered from rangeland ecosystems are cattle and sheep, and, to a lesser extent, goats. Invasive plant management can influence quality and quantity of forage, as well as its accessibility. Consequently, the quantity and quality of livestock products can be impacted, but the

Annual grasses significantly decrease forage capacity of rangeland because cattle typically avoid them once they begin to develop seedheads, which is usually late spring to early summer for cheatgrass. (Photo: Ryan Steineckert)

direction and degree of impact varies for the three classes of livestock based on forage preferences and the plant functional groups being managed. In addition, the longevity of control impacts varies dramatically among invasive weed management strategies and their efficacy.

Cattle. Most invasive weeds decrease forage production for cattle (Olson 1999). A substantial amount of literature shows an increase in forage as a response to invasive plant management over an untreated control, but experimental evidence showing a reduction in forage production with weed invasion is limited (except see Maron and Marler 2008). In addition to loss of forage, cattle tend to avoid areas with heavy infestations of weeds (Lym and Kirby 1987; Hein and Miller 1992). For example, leafy spurge (*Euphorbia esula* L.) reduces the carrying capacity of infested rangeland to near zero because cattle will not graze in areas with 10% to 20% cover of this weed. Few examples of overall economic costs of invasion have been published, but losses of forage for cattle on private land in California alone are estimated to be $7.65 million per year because of yellow starthistle (*Centaurea solstitialis* L.; Alison et al. 2007).

Among 60 articles addressing invasive weed management and forage, 17 indicated an increase in forage quantity, quality, or accessibility. Increases occur where desired species are sufficiently abundant to respond to control procedures (Kedzie-Webb et al. 2002). Increases in perennial grass biomass ranged from 10% (Lym and Messersmith 1990) to 1 935% (Masters et al. 1996) in response to weed control. Most commonly, weed control



using herbicides increased forage for cattle about two- to threefold after 3 yr (Sheley et al. 2000). For example, picloram used to control spotted knapweed (*Centaurea stoebe* L.) increased grass yield by 1 513 kg · ha^{-1} for 2 yr. Nearly all studies were 1 yr to 3 yr in duration, with the period of invasive plant control being about 2 yr or 3 yr. Very little is known about the long-term forage production after a single herbicide application or a sustained control program. However, Rinella et al. (2009) found that leafy spurge had increased and grasses decreased in comparison to nontreated areas 17 yr after picloram treatment.

Effective biological control only exists for a small portion of the total invasive weed species. However, biological controls have increased forage quality and quantity as well as accessibility for cattle where they do exist. Huffaker and Kennett (1959) reported large increases in availability of grasses and forbs as cattle forage 10 yr after the release of natural enemies of St. Johnswort (*Hypericum perforatum* L.). Increased grass production has been reported after release of biological control agents (Rees et al. 1996). *Longitarsus jacobaeae* (Coleoptera: Chrysomelidae) has reduced the density of tansy ragwort (*Senecio jacobaea* L.) and reduced cattle losses to pyrrolizidine poisoning to near zero (McEvoy et al. 1993; Coombs et al. 1996).

Sheep prefer grazing broadleaved plants and can be used to shift plant communities toward grasses that are preferred by cattle. Johnston and Peake (1960) used sheep to reduce leafy spurge basal area and increase the basal area of crested wheatgrass (*Agropyron desertorum* [Fischer ex Link] Shultes). Similarly, sheep grazing increased Idaho fescue (*Festuca idahoensis* Elmer) density and the frequency of Kentucky bluegrass (*Poa pratensis* L.), while reducing spotted knapweed (Olson et al. 1997). Livestock grazing has also been successfully used to reduce annual grasses growing among perennial grasses (Havstad 1994).

Science and technology have not advanced to the point that reseeding desired species is consistently successful, but seeding desirable plants into invasive plant–infested rangeland can increase the quantity and quality of forage for cattle (Enloe et al. 2005; Sheley et al. 2005). In one successful case, applying clopyralid plus 2,4-D in combination with streambank wheatgrass (*Elymus lanceolatus* Scribn & Sm.) was used to reclaim a rangeland heavily infested by Russian knapweed (*Acroptilon repens* [L.] DC.) to a stand dominated by the sod-forming grass (Benz et al. 1999). More recently, methods for repairing damaged ecological processes have increased the success of revegetation across highly variable landscapes (Sheley et al. 2006, 2009). In these studies, specific processes in need of repair were identified and modified to foster vegetation dynamics toward favorable speciesIntegrated invasive plant management strategies also have the potential to improve the quantity and quality of rangeland for cattle. Among 100 randomly selected studies investigating integrated invasive weed management, 65 indicated short-term positive responses in the quality and quantity of forage for cattle. In a few cases, the response was synergistic in favoring desired vegetation composition (Sheley et al. 2004; Jacobs et al. 2006). Additive and single main treatment effects were the dominant response, and in most cases nonnative grasses increased over nonnative target invasive weeds (Lym 1998; Endress et al. 2008).

Sheep and Goats. Because sheep and goats consume comparatively more forbs than grasses in their diets, invasive plant management does not benefit these small ruminants (Lym and Kirby 1987; Kronberg and Walker 1993). Most broadleaved weeds contribute to the forage quantity and quality of sheep and goats (Olson and Lacey 1994). Although the nutrient content of broadleaved invasive weeds varies with phenology, most are highly nutritious (Bosworth et al. 1980, 1985).

Conclusions and Management Implications. In general, the quantity and quality of cattle forage, and thus, cattle, are favored by weed management for a short period. Because many invasive weed management procedures increase forage yield for 2–4 yr, the benefits decrease with time following treatment. Sheep and goats prefer forbs as a major dietary component, and consume many weeds as quality forage. Broadleaved weed management has few positive benefits for sheep and goats. Invasive plant managers may maximize commodity production using multispecies grazing.

Control Undesirable Vegetation

Controlling undesirable vegetation on rangelands is difficult and rarely cost-effective. Compared to other land types, rangelands generate relatively low revenues per unit area. Typically, rangeland managers face expansive invasive plant infestations with few dollars for management. Additionally, invasive weeds tend to have high intrinsic growth rates and abundant seed production (Hobbs 1991; Rejmanek and Richardson 1996), which allow for rapid reinvasion of sites following use of herbicides, prescribed fire, and other invasive plant control practices (Lym and Messersmith 1985b; DiTomaso et al. 2006b). Therefore, when invasive plants are successfully controlled, they often reoccupy the area very quickly. The keys to sustainably controlling large rangeland weed infestations are frequent use of strategies that provide inexpensive, short-term control, such as prescribed grazing or infrequent use of more expensive strategies that provide longer-term control, such as restoration. In this section, we review widely used weed control strategies with an emphasis on their short- and long-term effectiveness, as well as their costs.

Prescribed Fire. Fire consumes weed standing crop, and in this sense fire consistently reduces undesirable vegetation. However, to have any lasting effect, prescribed fire must reduce the production of biomass in subsequent growing seasons, reduce the existing year's standing crop, and have a neutral or positive effect on desirable species**.** Some studies report increases in invasive weed biomass production due to fire (Young et al. 1972; Jacobs and Sheley 2003; Travnicek et al. 2005; Thacker et al. 2008), whereas others indicate decreases in biomass (Whisenant et al. 1984; DiTomaso et al. 1999). Whisenant et al. (1984) reported an extreme reduction of Japanese brome (*Bromus arvensis* L.) due to fire, which temporarily reduced this invasive annual grass by 85%. Conversely, Jacobs and Sheley (2003) found that fire more than doubled production of Dalmatian toadflax (*Linaria dalmatica* [L.] Mill. subsp. *dalmatica*). DiTomaso et al. (2006) concluded that the effects of fire depend, in part, on the weed's life history strategy (i.e., annual, biennial, perennial) and characteristics of the fire regime. Fire-based invasive plant management is complex and often not predictable, and detailed studies are needed to identify effective fire regimes for particular species or similar groups of species.

Prescribed fire in areas of invasion has varying effects on different species and can often lead to an increase in production of invasive species. (Photo: Clare Poulsen)

Applying prescribed fire is costly, so it is important for managers to carefully consider the longevity of control. Unfortunately, effects of fire on invasive plants are usually measured for only a year or two postburn. Cheatgrass (*Bromus tectorum* L.) has been measured for longer periods, but this weed does not appear amenable to fire-based control. On Western rangeland, cheatgrass tends to increase with fire because fire suppresses the less fire-tolerant competitors and available resources are rapidly acquired by cheatgrass (Young and Evans 1978; Vasquez et al. 2008). Cheatgrass invasion often increases fire frequency by increasing fine fuel loads, so burning cheatgrass can trigger a frequently repeated cycle whereby cheatgrass increases fire and fire increases cheatgrass (Knapp 1996). Conversely, prescribed fire that reduces invasive plants mainly destroys propagules, rather than by altering the environment in a manner that disfavors invasive weeds (DiTomaso et al. 2006a). When this is the case, weeds have only to replenish their propagule supplies to regain preburn abundances. Invasive plants tend to have high intrinsic growth rates which allow them to regain lost propagules quickly (Rejmanek and Richardson 1996; Grotkopp et al. 2002; Pyšek

and Richardson 2007). Therefore, prescribed fire will tend to provide only short-term control, and managers would need to burn regularly enough to maintain control for the long term. Finally, as with cheatgrass, there are many herbaceous perennials that cannot be controlled by fire alone, but fire can kill large quantities of surface-deposited seeds (Vermeire and Rinella 2009), so integrating fire with strategies that kill established plants may enhance control of some perennial invasive plants.

Herbicides. Herbicides are very useful for preventing small invasive plant infestations from producing seeds and spreading. They are also effective for controlling weeds during restoration projects so that seeded species have a better chance of establishing (Cione et al. 2002; Huddleston and Young 2005). These uses aside, it is generally not cost-effective to control large invasive plant infestations with herbicides alone because the repeated applications required to maintain control (every 1–3 yr) are too expensive (Lym and Messersmith 1985b; Sheley et al. 1998; Young et al. 1998). Controlling rangeland invasive plants rarely increases forage production enough to offset the herbicide costs (Griffith and Lacey 1991; Bangsund et al. 1996). Furthermore, invasive annual grasses often proliferate after herbicides kill associated invasive forbs, so controlling invasive broadleaved forbs often just replaces undesirable forbs with undesirable annual grasses (Shinn and Thill 2003).

Another problem with large-scale herbicide treatments is that they often kill associated native forbs and shrubs (Erickson et al. 2006; Sheley and Denny 2006; but see, Rice et al. 1997a). Whereas invasive weeds usually recover from herbicide quickly, a recent study of leafy spurge shows desired native plants can fail to recover from herbicides regardless of the length of the recovery period (Rinella et al. 2009). In that study, leafy spurge filled niches left vacant after herbicides removed native plants. Paradoxically, when herbicides damage native plants, invasive plants may ultimately become more abundant in response. Although herbicides play a critical role in weed prevention and restoration, the scientific literature causes us to question their use as stand-alone tools for controlling expansive invasive plant infestations.

Prescribed Grazing. Prescribed grazing encourages the targeted use of invasive plants by manipulating timing, intensity, and frequency of herbivory and selecting animal classes based on their dietary preferences. For example, goats prefer trees and shrubs and forbs compared to grasses, so they are sometimes stocked on grasslands invaded by pines and junipers or invasive forbs, such as knapweed (Campbell et al. 2007).

The key difference between prescribed grazing and other invasive plant management strategies is that it can be affordably used on an annual basis to reduce invasive plant standing crop and biomass production in many situations. Sheep and goats can be economically profitable in well-managed operations or they can serve as additional revenue sources in cattle operations (Williams et al. 1996; Bangsund et al. 2001). Additionally, using sheep or goats to graze and reduce exotic forb standing crop can increase the amount of forage accessible to cattle and increase overall forage utilization (Lym and Kirby 1987).

Desirable rangeland species generally increase after land management practices reduce invasive plant biomass (Lym and Messersmith 1985a; Belcher and Wilson 1989; Sheley et al. 2000). As a consequence, prescribed grazing is more worthwhile when it reduces subsequent invasive plant biomass in addition to reducing standing vegetation. Invasive plant biomass responses to grazing depend on the timing, intensity, and frequency of grazing, as well as the class of livestock. For example, three studies reported no effect of sheep grazing on leafy spurge production because most grazing occurred during and after leafy spurge seed production, and at only a single time during the season (Lacey and Sheley 1996; Olson and Wallander 1998; Seefeldt et al. 2007). Alternatively, four other studies reported fairly consistent declines in leafy spurge over time, and in these studies, grazing occurred multiple times prior to seed production (Johnston and Peake 1960; Lym et al. 1997; Jacobs et al. 2006; Rinella and Hileman 2009). Controlling invasive plants using livestock requires the development of relatively complicated,

“Although herbicides play a critical role in weed prevention and restoration, the scientific literature causes us to question their use as stand-alone tools for controlling expansive invasive plant infestations.”

Prescribed grazing and biological control can be effective treatments when implemented into an integrated management program. (Photo: Sharon Bingham)

strategically designed strategies for each species based on their tolerance and/or resistance to herbivory. Detailed investigations are needed to identify prescribed grazing strategies that are effective for specific invasive species in particular environments.

Biological Control. Biological control agents can clearly damage individual plants (Pecinar et al. 2007; Thomas and Reid 2007; Zalucki et al. 2007), but unfortunately, these effects often fail to cause appreciable reductions in undesirable vegetation (DeLoach 1991). Many agents are released in hopes that one or a combination of them will prove effective (McEvoy and Coombs 1999). Unfortunately, this lottery approach is rarely effective in controlling large populations. Many invasive plants remain highly problematic despite being the target of many releases of biological control agents for decades (Zalucki et al. 2007; Story et al. 2008). Furthermore, the risks of deleterious off-target effects increase with the number of releases (Louda et al. 2005; Pearson and Callaway 2005).

Risks and failures aside, biological control is occasionally extraordinarily successful against invasive weeds. The most-cited examples include two introduced beetles that reduced St. Johnswort density by greater than 99% in much of its introduced range (Harris and Maw 1984), and three insects that substantially reduced ragwort in western Oregon (McEvoy et al. 1991; Denslow and D'Antonio 2005). Although these are the best-studied examples, other rangeland weed species have been targeted, with quite varied results, including recent introductions that show promise for

controlling salt cedar (*Tamarix ramosissima* Ledeb.; Hudgeons et al. 2007). In those few special cases, biological control has the unique benefit of providing relatively inexpensive partial weed control over expansive areas for indefinite periods of time.

Mechanical Control and Seeding.

Mechanical methods of herbaceous weed control include tillage and mowing. Mechanical control treatments of tillage and mowing can cause substantial reductions in invasive plant standing vegetation, but they are only truly effective when future biomass production is reduced. Invasive plant responses to mowing have been mixed, with some studies reporting appreciable decreases in weed biomass production (Benefield et al. 1999; Rinella et al. 2001) and other studies reporting no detectable change (Benz et al. 1999; Renz and DiTomaso 1999). Collectively, studies indicate the responses of undesirable vegetation to mowing depend on species, timing of mowing, and other factors. Finally, because invasive weeds quickly recover after mowing is discontinued, mowing must occur frequently to provide continuous control.

Tillage alone can be used to control invasive plants on rangeland under some circumstances. In one study, tillage alone provided no sustained control of perennial pepperweed (*Lepidium latifolium* L.; Young et al. 1998), and in another study, repeated tillage prior to a serious frost reduced leafy spurge well (Lym and Messersmith 1993). Invasive weeds tend to recover quickly when tillage is discontinued.

In addition to controlling invasive plants, tillage provides safe sites for seeded species, and some studies report that competition from seeded species has partially controlled undesirable vegetation (Lym and Tober 1997; Bottoms and Whitson 1998; Sheley et al. 2001; Thompson et al. 2006). However, other studies have reported that seeded species provided no weed control (Sheley et al. 1999; Mangold et al. 2007). In the latter case, it is possible data were collected before the seeded species grew large enough to compete with the reemerging invasive plants. Theoretically, when seeded species develop self-sustaining populations, these populations should suppress undesirable vegetation indefinitely through resource competition. Therefore, despite the lack of evidence, there are likely to be distinct advantages to integrating seeding with other practices that provide only short-term control, such as herbicides and tillage.

Conclusions and Management Implications.

Many questions remain regarding control of invasive plants, and many of these questions pertain to inconsistencies in the responses of undesirable vegetation to various controls. For example, there are cases in which individual treatments, such as herbicides, grazing, or fire have reduced invasive weeds. There are also cases in which treatments have failed to alter the abundance of invasive plants, and there is even some evidence that invasive weed control occasionally increases weed species. Furthermore, except for a few situations using biological control or annual repeated grazing, it is often questionable whether or not individual invasive weed control strategies are worthwhile because the control they provide is so ephemeral and expensive.

Presumably, integrating multiple control strategies should lead to more consistent, longer-lasting suppression of invasive plants. But integrated strategies are more costly, and there is still no guarantee that the level and longevity of invasive plant control will be satisfactory (Sheley et al. 2001; Lym 2005). Much research is needed to identify affordable, consistently effective strategies for controlling undesirable vegetation.

Create a Desired Plant Community

On most sites, the species that invasive plants suppress or displace comprise both nonweedy exotic species and natives (Enloe et al. 2007). Some of the suppressed natives and nonnatives are often valuable forage plants, and increased forage production often provides the impetus for controlling invaders (Lym and Messersmith 1985a). In many cases, controlling undesired species does not lead to a desired plant community. A variety of other objectives may also be met by creating desired plant communities including increasing native species diversity, increasing habitat for wildlife, improving soil and water quality, and reducing reinvasion. In this section, we investigate how desired, especially native, species respond to invasive weed control and examine efforts

A desired plant community should include a variety of species that fill multiple niches which reduces the likelihood of reinvasion. (Photo: Brett Bingham)

“These studies suggest that prescribed grazing may have potential for restoring desired species, but invaded communities are likely to quickly regress to their pregrazing weedy state when prescribed grazing is discontinued.”

to reestablish desired species from seeds. The choice of desired species depends upon management goals. In this document, the goal is considered to be to establish and maintain a healthy, functioning plant community that is resistant to invasion and meets other land use objectives (Sheley et al. 1996).

Prescribed Grazing. To our knowledge, only two studies have provided detailed assessments of plant community responses to prescribed grazing of invasive plant–infested rangeland. Olson and Wallander (1998) studied responses of a leafy spurge–infested plant community to prescribed sheep grazing. The authors concluded that grazing reduced leafy spurge stem height without affecting stem density, so grazing presumably lowered leafy spurge biomass production. Grazing increased the density and frequency of several native (Idaho fescue, western wheatgrass [*Pascopyrum smithii* {Rydb.} A. Löve], Sandberg bluegrass [*Poa secunda* J. Presl]) and nonnative (Kentucky bluegrass, annual bromes) grasses, and decreased density of a nonnative dandelion (*Taraxacum officinale* Weber). In a similar study, on spotted knapweed-infested rangeland, Olson and Wallander (1997) found that sheep grazing reduced spotted knapweed rosette and adult plant density. As in the leafy spurge study, grazing increased density and frequency of native Idaho fescue and nonnative Kentucky bluegrass. The native forb arrowleaf balsamroot (*Balsamorhiza sagittata* [Pursh] Nutt.) was not influenced by grazing. These studies suggest that prescribed grazing may have potential for restoring desired species, but invaded communities are likely to quickly regress to their pregrazing weedy state when prescribed grazing is discontinued. To be successful, prescribed grazing will likely need to be carried out indefinitely. It is unfortunate that so few studies have evaluated native species responses to prescribed grazing of weed-infested rangeland.

Biological Control. Denslow and D’Antonio’s (2005) review of the literature clearly demonstrates that successful biological control of rangeland invaders can, but does not always, have positive effects on suppressed desired species. A classic successful example is the control of St. Johnswort by the leaf-beetles *Chrysolina quadrigemina* Suffrian and *Chrysolina hyperici* Forster in California rangeland. Within 5–10 yr of leaf-beetle introduction, St. Johnswort had virtually disappeared from sites in several California counties (Huffaker and Kennett 1959). It was reduced to less than 1% of its prerelease cover, and replaced by a combination of native and exotic grasses, which greatly increased available forage. Similarly, a combination of several insects reduced ragwort to a fraction of its former abundance in several California sites (Pemberton and Turner 1990), and throughout western Oregon (McEvoy et al. 1991). In both situations, control agents persisted, ragwort remained under control for more than a decade, and desired species responded favorably. More recently, a suite of biological control agents led to successful control of diffuse knapweed (*Centaurea diffusa* Lam.) at sites in Colorado, Montana, Oregon, Washington, and British Colombia (Myers 2004; Smith 2004; Seastedt et al. 2007), and successful or partial control of spotted knapweed in sites in Colorado and Montana with some responses by desired species (Story et al. 2006; Seastedt et al. 2007). Similarly, several flea beetles (*Apthona* spp.) have displayed variable, but sometimes quite successful control of leafy spurge and subsequent release of desired species in South Dakota, North Dakota, and Montana (Larson and Grace 2004; Butler et al. 2006; Cornett et al. 2006). Additionally, Lesica and Hanna (2004) provide an example of positive native plant community responses to biological control.

Herbicides. Rangeland herbicides tend to selectively kill either grasses or forbs. Therefore, native grasses are typically not damaged by the herbicide used to control invasive forbs. In fact, many native grasses increase following herbicide control of invasive forbs (Sheley et al. 2000; Laufenberg et al. 2005; Sheley and Denny 2006). Similarly, native forbs often increase after herbicides kill invasive grasses (Cione et al. 2002; Wilcox et al. 2007).

Herbicides are sometimes used to control invasive grasses even though herbicide-sensitive native grasses are present (Kyser et al. 2007). Likewise, herbicides are used to control invasive forbs growing with native forbs and shrubs (Fuhlendorf et al. 2002). Sometimes native species escape extensive damage by herbicides

or are able to quickly recover from damage. For example, when the broadleaf herbicide picloram was applied to spotted knapweed–infested rangeland during the summer-dormant period of most native forbs, Rice et al. (1997b) found the herbicide had only mild transient effects on native forbs. Similarly, Erickson et al. (2006) found that herbicide control of leafy spurge with quinclorac did not damage the threatened prairie fringed orchid, although imazapic damaged the orchid. Also, Simmons et al. (2007) found the "nonselective" herbicide glyphosate provided substantial short-term control of an invasive grass while damaging native grasses to a lesser extent or not at all. Finally, Barnes (2007) found that grass-specific herbicides promoted native warm-season grasses by reducing abundance of the exotic grass, tall fescue.

In contrast to these examples, there are other cases demonstrating extensive herbicide damage to natives. For example, Sheley and Denny (2006) concluded that any of three herbicides used to control sulfur cinquefoil (*Potentilla recta* L.) continued to suppress native forbs 2 yr after application. Similarly, Shinn and Thill (2004) found that imazapic, a herbicide that is active against invasive annual grasses, substantially injured native perennial grasses as well. Herbicide damage to native species would not be a problem if the natives consistently recovered, but a recent study showed that herbicide control of rangeland weeds can pose very serious long-term threats to native forb populations (Rinella et al. 2009). In addition to sometimes extensively damaging native species, herbicides are expensive and they generally provide only short-term weed control. Therefore, herbicides alone are unlikely to create desired native plant communities. However, herbicides are critical in preventing spread of small weed infestations and in integrated weed management.

Prescribed Fire. Prescribed fire can boost native species and reduce populations of annual invaders by consuming their seeds (DiTomaso et al. 2006b). For example, Harmoney (2007) found that fire greatly reduced Japanese brome and increased two native grasses above an unburned control. Also, fire reduced invasive yellow starthistle and three invasive annual grasses while greatly increasing diversity and species richness of native forbs (Hastings and DiTomaso 1996; DiTomaso et al. 1999). However, depending on fire timing, species identity, and other factors, fire can also boost invaders (Young et al. 1972; Jacobs and Sheley 2003; Travnicek et al. 2005; Thacker et al. 2008) and cause severe damage to native populations, such as big sagebrush (*Artemisia tridentata* Nutt.) growing with cheatgrass (Young and Evans 1978; Knapp 1996).

To our knowledge, no studies have measured long-term invasive weed and native species responses to prescribed burning. Unfortunately, favorable responses likely will be short-lived. Weeds generally reinvade very quickly following fire (Young and Evans 1978), and native plants are likely to revert to their suppressed preburn state following reinvasion. It is unlikely that managers can burn vegetation regularly enough to maintain native populations, but is possible to integrate fire with other strategies in hopes of providing longer-term restoration of native species (MacDonald et al. 2007).

Mowing. A small number of studies have evaluated desired plant responses to mowing of invasive weed–infested rangeland. Wilson and Clark (2001) found mowing invasive grass-infested rangeland for 4 yr greatly restored native prairie grasses. Similarly, MacDougall and Turkington (2007) found that 5 yr of mowing at the time of invasive grass flowering shifted the plant community toward desired forbs and grasses. In contrast to these successes, Simmons et al. (2007) found mowing had little or no detectible effect on an invasive perennial grass and native species, and Brandon et al. (2004) reported mowing increased invasive forb abundances. Collectively, these studies indicate native species responses to mowing depend on the relative susceptibility of desired species and invaders to different timings, heights, and frequencies of defoliation as well as other factors. Finally, when beneficial mowing regimes are identified, they will likely have to be carried out indefinitely or combined in an integrated management strategy to maintain native species because invasive weeds tend to recover quickly when mowing is discontinued.

Seeding. In revegetation projects, the desired plant community is in large part dictated by the species in the seed mix. A few

"It is unlikely that managers can burn vegetation regularly enough to maintain native populations, but is possible to integrate fire with other strategies in hopes of providing longer-term restoration of native species."

“It has become clear that nonnative grasses have tended to outperform native grasses in revegetation studies.”

revegetation efforts have resulted in fairly diverse assemblages with impressive stability (Blumenthal et al. 2003; Martin et al. 2005; Martin and Wilsey 2006; Foster et al. 2007). However, these successes were obtained from areas with relatively innocuous weed species, not highly tenacious rangeland invaders such as leafy spurge and spotted knapweed.

Where invaders dominate rangelands, managers often strive for the humble goal of introducing one or a few desired or native species in hopes they will provide forage, wildlife habitat, and other values. Seeding is very expensive, so successful seedling establishment is critical. Seedling establishment is most effective when seeding has been integrated with herbicides and/or tillage (Lym and Tober 1997; Ferrell et al. 1998; Whitson and Koch 1998; Masters et al. 2001; Huddleston and Young 2005; Simmons 2005), but establishment failures are common (Lym and Tober 1997; Masters et al. 2001; Sheley et al. 2001; Wilson et al. 2004). Seedling establishment depends on a myriad of factors, not the least important of which is interannual climatic variation (MacDougall et al. 2008), which cannot be controlled or effectively predicted. Beyond establishment, seeded species must maintain viable populations over the long term for seeding to be worthwhile. Unfortunately, desired native species are rarely measured for more than a few years after sowing into invaded rangelands. Among the longer-term investigations, Bottoms and Whitson (1998) found that seeded thickspike wheatgrass (*Elymus lanceolatus* [Scribn. & J.G. Sm.] Gould) and western wheatgrass maintained healthy stands that suppressed Russian knapweed 5 yr after seeding. Similarly, Ferrell et al. (1998) found that seeded wheatgrasses produced large quantities of biomass and continued suppressing leafy spurge 6 yr after seeding. Also, a mixture of four native warm-season grasses contributed substantially to biomass production 6 yr after being sown into spotted knapweed–infested plots (MacDonald et al. 2007). To our knowledge, Ferrell et al. (1998) document the longest-term measurements of seeded species in invasive plant–infested rangeland. The authors found that two grass species remained fairly abundant and partially suppressed leafy spurge 10 yr after seeding. It has become clear that nonnative grasses have tended to outperform native grasses in revegetation studies (Ferrell et al. 1998; Asay et al. 2001; Sheley et al. 2001).

Given the expense of seeding rangelands, the lack of long-term measurements is troubling. Future restoration research should focus on determining whether or not native species can persist with invasive species beyond a few years after seeding. To some extent, restoration efforts are predicated on the assumption that improper land management causes weed invasions. If proper management does not prevent invaders from dominating the original community, then we should not expect proper management to prevent invaders from dominating the restored community. It is sobering to consider that rangeland restoration may be doomed to fail over the long term wherever invaders have displaced natives despite good range management.

Conclusions and Management Implications.

Many studies provide no information on desired or native species responses to weed management, and a few others provide only cursory information. Biological control is relatively inexpensive to implement once developed, and a few biological control programs have restored natives to an impressive extent. However, biological control sometimes fails completely, and it has many risks. More research is needed to elucidate the risks and benefits of biocontrol. Desired and native species responses to prescribed grazing have been limited, but two studies indicate that annually applied prescribed grazing can shift plant communities toward a desired state. Herbicides often control target invaders while allowing associated species to increase in abundance. However, herbicides are quite expensive and herbicides alone provide only short-term weed control. Furthermore, herbicides pose considerable risks to some desired species. Herbicides are useful for preventing spread of weed infestation, and thus helpful in maintaining a desired plant community that has not been invaded. Prescribed fire can be beneficial to desired species, but it can also harm natives and increase invasion. Effective prescribed fire regimes will likely need to be repeated regularly, and can be used occasionally to restore desired plant communities prior to invasion to help

keep them resistant. Effective mowing will probably need to be carried out often to be successful, and in some cases, the plant material may need to be removed. A few studies have reported that seeded native species remained abundant and suppressed tenacious rangeland invaders 5 yr or 6 yr after seeding. This is promising, but longer-term measurements are desperately needed to determine if the benefits of seeding warrant its high costs.

Change Underlying Causes of Weed Invasion

The benefits of invasive plant control depend in large part on the longevity of control and the resulting desired plant community. In turn, this depends on the ecological causes underlying the original invasion and our ability to alter those causes in favor of desired species. Because invasive species are rarely eradicated, if the original causes of invasion are not repaired, reinvasion is likely (Sheley and Krueger-Mangold 2003). Furthermore, given the large areas and low economic returns per unit area typical of rangeland, temporary invasive weed control is rarely economically sustainable. In this section, we consider the durability of invasive species control and desired community restoration.

Durability Depends on Original Cause of Invasion. Invasion is often the result of changes to an ecosystem that inhibit native species, and thereby reduce the competition faced by invasive species (Facon et al. 2006). Perhaps the most important barrier to invasion is the presence of desired species. Desired species garner much of the water, nitrogen, light, and other resources that would otherwise be available to invaders. This resistance to invasion is often described as "biotic resistance" (Maron and Vila 2001; D'Antonio and Thomsen 2004; Levine et al. 2004), and depends a great deal on environmental conditions (Shea and Chesson 2002). Because native species are, by definition, adapted to historical environmental conditions (Landres et al. 1999), changes to these conditions are likely to make them less well-adapted, and less able to resist invasion. Consequently, changes in environmental conditions appear to be a common cause of invasion (Daehler 2003; Facon et al. 2006). These changes can be dramatic, such as soil tillage or improper grazing, or subtle, such as nitrogen deposition or loss (Vasquez et al. 2008). Where such changes underlie invasion, the key question facing managers is whether the change can be reversed. Among the many underlying causes of invasion, managers have had the most success reversing the following three: past disturbances, reversed via successful restoration; returning grazing to systems, and enemy release, through biological control.

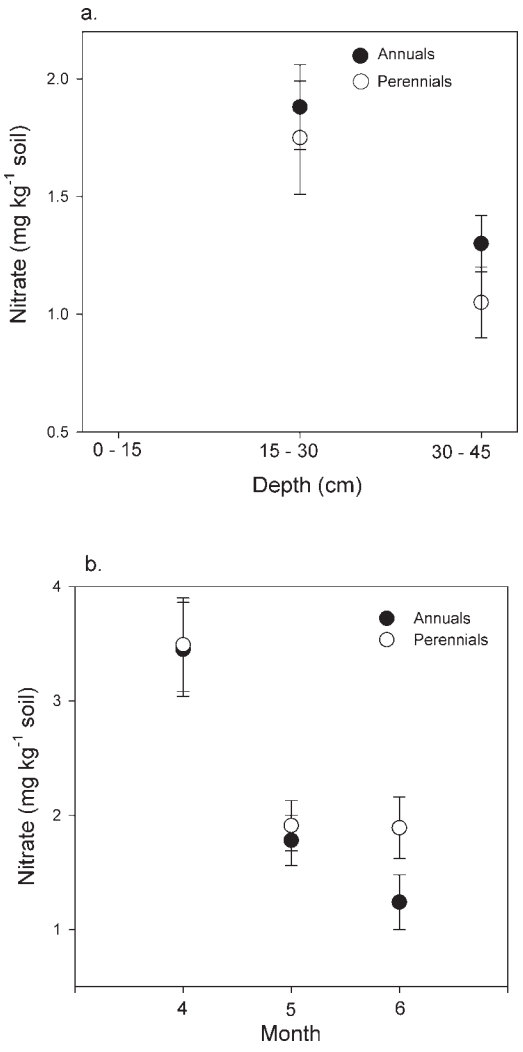


FIGURE 3. a, Exotic annual species (●) are less effective than native perennial grasses (○) at extracting soil nitrate at all depths in May. **b**, Monthly sampling was conducted at the 0- to 15-cm depth. Error bars = 1 SEM; $n = 32$.

Restoration as a Long-Term Solution to Previous Novel Disturbances. A key question for determining the likelihood of long-term invasive plant control is the degree to which a particular invasion is caused by a novel disturbance—a disturbance to which native species are not well adapted. Where such disturbances underlie the invasion, and can be prevented in the future, it is much more likely that a stable, desired plant community

> …reestablishing proper grazing regimes that maintain vigorous plants and healthy plant communities can limit invasion."

can be restored. A combination of invasive weed control (typically with herbicides) and restoration can potentially reestablish native species. Long-lived perennial species often garner much of the water, nitrogen, and other resources available in grassland ecosystems (Wedin and Tilman 1990; Tilman and Wedin 1991; Baer et al. 2002; Seabloom et al. 2003; Fig. 3). Consequently, the presence of perennial species can provide substantial biotic resistance against invasion (Blumenthal et al. 2003, 2005; Seabloom et al. 2003; Bakker and Wilson 2004; Levine et al. 2004). Where disturbance damages or removes resident species, it can increase resource availability and therefore provide opportunities for invasive species (Hobbs and Huenneke 1992; Davis et al. 2000). However, where desired or native species can be restored, their presence may be sufficient to reduce resource availability and keep invasions from recurring (Blumenthal et al. 2003; Seabloom et al. 2003).

Few studies have actually tested whether restoration leads to persistent invasive plant control. Ferrell et al. (1998) studied the response of leafy spurge to herbicides, tillage, and then seeding of several native and nonnative grass monocultures. Five years postseeding, one grass was very rare, but the others substantially suppressed leafy spurge. The two most effective species (which were nonnative) were reassessed 10 yr postseeding, and these species continued to suppress leafy spurge. Similarly, Bottoms and Whitson (1998) found that herbicides and tillage followed by seeding of several native and nonnative grasses greatly suppressed Russian knapweed 5 yr postseeding. Herbicide, tillage, and seeding of native tallgrass prairie species greatly suppressed weeds in an old field 7 yr after seeding (Blumenthal et al. 2003, 2005). Although most of the weed species in this study are not considered aggressive invaders of rangeland, two of the species inhibited by restoration can be desired species or invaders: Kentucky bluegrass and smooth brome (*Bromus inermis* Leyss.). Finally, Seabloom et al. (2003) found that 5 yr after restoration, native perennials comprised the majority of the plant biomass in an area otherwise dominated by exotic annuals. These studies suggest seeding may sometimes provide cost-effective, long-term weed control. However, additional longer-term measurements are desperately needed to better evaluate the long-term benefits of seeding.

Altering Disturbance Regimes as a Solution to Changing Environmental Conditions. Where invasion is caused by past disturbance, long-term weed control and restoration may often be achieved with a combination of invasive weed control and seeding. Where invasion is caused by ongoing disturbance, however, or where past changes have led to new stable states, it may also be necessary to change the disturbance regime (Suding et al. 2004). The most important example of this problem in North American rangeland ecosystems is the increase in fire frequency that is both caused by and helps to perpetuate cheatgrass invasion (D'Antonio and Vitousek 1992). Because shrub-steppe ecosystems are not well adapted to frequent fire, the new disturbance regime appears to preclude successful restoration. Consequently, fire suppression is required to allow native species to compete against cheatgrass and other fire-tolerant invaders (Brooks et al. 2004). Fire suppression may or may not be sufficient to allow native species to compete against cheatgrass. If cheatgrass invasion is caused by a combination of fire and past disturbance, then fire suppression and restoration may be sufficient. If cheatgrass invasion is also driven by ongoing changes, such as nitrogen deposition or the amount/timing of grazing, then fire suppression and restoration may not be sufficient. In contrast to shrub-steppe ecosystems, tallgrass prairie evolved with frequent fire. There, the absence of fire can lead to invasion, and prescribed fire can be a long-term solution to invasion (Smith and Knapp 1999; Copeland et al. 2002). Similarly, grazing can be a disturbance that either favors or inhibits invasive plants, depending on the grazing history of the site (Mack and Thompson 1982; Milchunas et al. 1988, 1992; Bock et al. 2007).

Grazing as a Cause and Solution to Invasion. Invasive species have evolved with grazing animals, creating a complex relationship among grazing preferences and plants' abilities to resist and tolerate grazing (Heitschmidt and Stuth 1991). Invasion can increase when competitive, desired species are defoliated beyond their ability to recover by the following growing season (Sheley et al. 1997).

Thus, reestablishing proper grazing regimes that maintain vigorous plants and healthy plant communities can limit invasion (Sheley et al. 2008). Similarly, lack of grazing of invaders can stimulate invasion. Since goats and sheep dominate most grazing in areas where many invasive weeds evolved, introducing them into areas with serious infestations of these species can reduce invasive weeds (Olson 1999).

Biological Control as a Solution for Invaders that have Escaped Natural Enemies. It is also possible that invaders succeed, overcoming biotic resistance, without any change in environmental conditions. Invaders may have an advantage over native species for other reasons. For example, the enemy release hypothesis proposes that invasive plants have an advantage over native species because they have escaped natural enemies when introduced to a new range (Maron and Vila 2001; Keane and Crawley 2002; Mitchell and Power 2003; Colautti et al. 2004; Blumenthal 2006). When enemy release is driving invasion, biological control, reuniting invasive species with specialized enemies, may be the most durable control method (Fig. 4). The reversal of enemy release by biological control is inexact. Only a subset of the enemies from the original range is introduced. The introduced enemies, however, are often missing their own predators, and may influence the invasive weed more strongly than they would have in their native range (Keane and Crawley 2002).

Long-Term Solutions When Causes of Invasion Cannot Be Reversed. All of the above sections discuss situations in which reversing the cause of invasion should help native species compete effectively against invasive species. However, not all causes of invasion can be reversed. For example, if atmospheric CO_2 enrichment causes invasion of otherwise healthy native plant communities (Smith et al. 2000), there may be no way to manage the environment to enable the native plant community to resist invasion. Other types of global change, such as nitrogen deposition and altered precipitation, may lead to similar problems (Dukes and Mooney 1999; Brooks 2003; Vila et al. 2007; Blumenthal et al. 2008). In such situations, efforts to increase biotic resistance, and therefore achieve long-term weed control, may require the use of plant species from outside the local plant community (Seastedt et al. 2008). There may also be situations in which long-term solutions simply do not exist. If invasive species are better adapted to local conditions than are desired species and the local conditions cannot be altered, management choices may be limited to relatively expensive ongoing control or learning to manage the invasive species as novel ecosystems. Given the frequency of novel disturbances, opportunities for grazing management, and enemy release, it appears possible that the causes of invasion could be reversed in many cases. Where possible these approaches are likely to lead to more persistent invasive weed control and more stable desired plant communities.

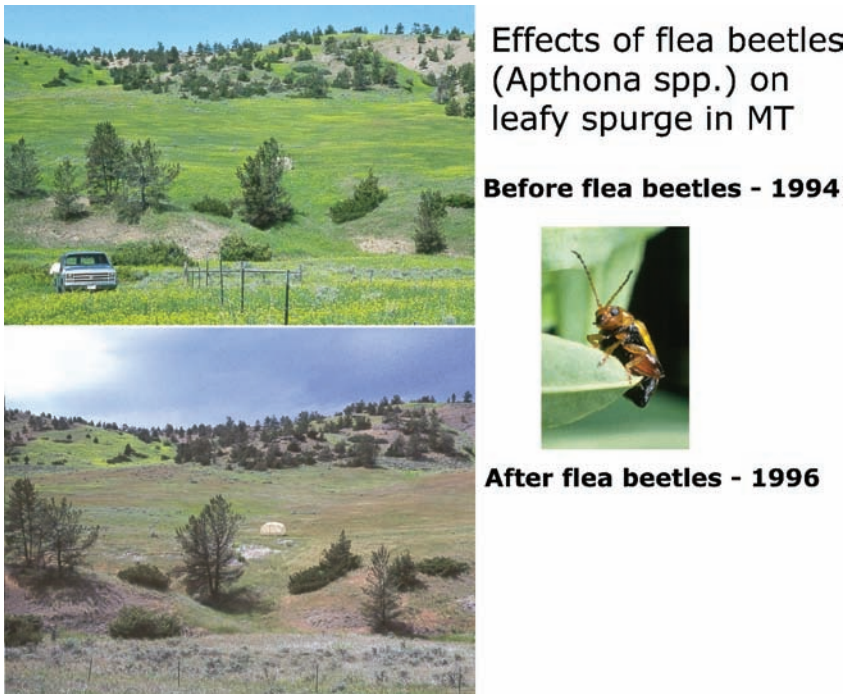


FIGURE 4. Flea beetle (Apthona spp.) (right photo) impact on leafy spurge in Montana (bottom photo; 1996) 3 years following their introduction in 1994 (top photo).

Restore Desired Vegetative Cover to Protect Soils, Control Erosion, Reduce Sediment, Improve Water Quality and Quantity, and Enhance Stream Flow

Plant community structure and composition are important drivers of ecosystem function and services, including protection and conservation of soil and water resources (Chapin et al. 2000). Our ability to reestablish desired vegetation on invasive plant–infested

Invasive species can have a significant impact on soil resources by altering plant cover, litter inputs, and the amount and distribution of bare ground. (Photo: Alex Boehm)

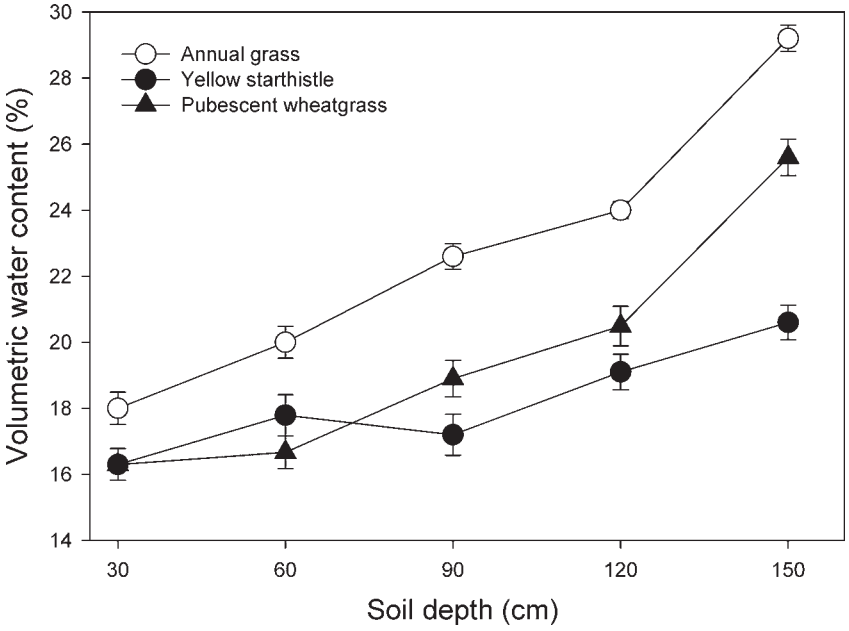
rangeland is extremely limited, particularly in areas with low precipitation. The proportions of plant cover and bare ground are generally the most important factors that determine the degree to which soil and water resources are protected and conserved by a plant community. To this end, when desired vegetation has been reestablished on degraded rangeland with low total vegetative cover and a high proportion of bare ground, protection and conservation of soil and water resources generally increase (Pyke et al. 2002; Pierson et al. 2007a). The bulk of restoration programs, however, have not evaluated if reestablishment of desired vegetation has effectively impacted soil or water resources. Of the few synthetic efforts to date on rangeland, there is little evidence to suggest that reseeding efforts are successful in establishing enough plant cover to significantly improve protection of soil and water resources beyond what is attained through natural site recovery processes (Pyke et al. 2003; Byers 2004). Of equal importance is the notion that weedy plant communities and desired plant communities may not necessarily differ in their ability to protect and conserve soil and water resources. In some instances, weedy plant communities may be important in rapidly stabilizing heavily disturbed communities on steep slopes, preventing loss and damage to soil and water resources (Pierson et al. 2007b). Nevertheless, there are a number of instances where restoring desired vegetation cover in weed-infested communities may benefit soil and water resources. Although empirical data are limited, some general principles have emerged that should allow reasonable prediction of when restoring desired vegetative cover on weed-infested rangeland may achieve these benefits.

Soil Resources. Although a few studies show that establishing a desired vegetation cover on weed-infested rangeland can increase protection of soil resources (Lacey et al. 1989), the bulk of evidence is largely observational (Sperber et al. 2003), limiting our ability to develop generalities. However, processes and factors associated with effective soil conservation are fairly well defined, which may allow relatively accurate predictions of when restoration of desired vegetative cover will provide soil conservation benefits. Plant cover as well as the proportion and connectivity of bare ground are central factors determining erosion and sediment yield. More canopy cover lowers the effective energy of raindrops as well as the amount of soil exposed to rainfall impact (Blackburn et al. 1994). Large, interconnected patches of bare ground concentrate runoff and increase flow velocities and erosion (Schlesinger et al. 1990; Pierson et al. 2007a). The evidence suggesting that weeds alter soil physical properties is mixed, and appears species-specific (Sperber et al. 2003; Norton et al. 2004). The most significant impacts weeds have on soil resources are related to the degree to which weeds affect plant cover, litter inputs, and the amount and distribution of bare ground (Lacey et al. 1989; Pierson et al. 2007a). Therefore, soil conservation benefits may be achieved if restoring desired vegetation on weed-infested rangeland increases plant cover and/or decreases connectivity of bare patches and plant interspaces. On the other hand, if restoration efforts do not significantly alter these parameters, then establishing desired vegetation may not significantly improve conservation of soil resources.

Water Resources. The large negative effect of invasive plant species on water resources and the conservation benefit achieved by restoring desirable species is partially documented in rangeland riparian systems (Zavaleta 2000; Shafroth et al. 2005). However, on upland systems the benefits of restoring desired species on weed-infested rangeland are less well studied and the effects more nuanced. Several case studies examine weed effects on water resources on uplands and there have been extensive studies on individual plant water use patterns (Lambers et al. 2000; Enloe et al. 2004; Kulmatiski et al. 2006). Therefore, although the data on hand are limited, this information can be used to develop some general predictions as to the effects of weeds on water resources as well as the conservation benefits that may be obtained by establishing desired species on weed-infested rangeland. Patterns and rates of plant water use are determined by plant size, phenology, rooting depth, and root densities (Lambers et al. 2000). Weeds that differ significantly from desired vegetation in these traits have the potential to alter the pattern and amount of water available on rangeland. For example, annual grasses that have invaded sagebrush steppe systems initiate growth and use water earlier in the growing season compared to the native perennial bunchgrasses (Kulmatiski

et al. 2006). As a consequence, these weedy plant communities extract water at a faster rate earlier in the growing season than the desired plant community. In this case, eliminating the weeds and restoring desired perennial plants may allow water to remain available to plants for a longer duration during the growing season. As a contrasting example, deep-rooted forbs that invade native bunchgrass communities can deplete soil water at greater depths later in the growing season compared to bunchgrasses (Enloe et al. 2004; Fig. 5). In this case, restoring desired bunchgrasses may help conserve deep soil water.

FIGURE 5. Volumetric soil water content (%; mean ± SE) by depth averaged across time of sampling and year. Plant community by soil depth comparisons: yellow starthistle vs. annual grasses ($F = 3.33$, $P = 0.0262$), yellow starthistle vs. pubescent wheatgrass ($F = 0.27$, $P = 0.8738$).



Conclusions and Management Implications.

Our ability to restore desired vegetation on arid and semiarid rangeland is limited. Restoring desired species may not always result in a conservation benefit in terms of soil and water resources. Although few studies have examined the conservation benefits of establishing desired species, basic knowledge about soil stability and hydrological processes allows reasonable prediction of scenarios where restoring desired species will benefit soil and water resources. Namely, if restoring desired species increases cover or litter inputs and/or decreases the amount or continuity of bare ground, a soil and water conservation benefit will likely be achieved. The impact of weedy plants on water resources and the benefits achieved by restoring desired species will mainly depend on the degree to which these species groups differ in size, phenology, rooting depth, and root densities. When these species groups exhibit large differences in one or more traits, substantial conservation benefits may be achieved. On the other hand, when these differences are small the conservation benefits may be negligible.

Maintain or Enhance Wildlife Habitat Including that Associated with Threatened and Endangered Species

Invasive plant species often change ecosystem structure and function, directly impacting wildlife habitat (DiTomaso 2000; Masters and Sheley 2001). It is not surprising, therefore, that invasive species removal has been shown to benefit wildlife in a number of systems. Relationships between invasive plant species and wildlife, however, are often more complicated, involving both positive and negative effects. For example, saltcedar (*Tamarix* spp.) can provide suitable habitat for the endangered southwestern willow flycatcher (*Empidonax traillii extimus* Nelson) and other avian species that nest in midcanopy vegetation, but poor habitat for many other avian species (Dudley and DeLoach 2004; Shafroth et al. 2005; Brand et al. 2008; Fig. 6). The net benefit of invasive species control on wildlife habitat not only depends on the balance between negative and positive effects of invasive plants on wildlife habitat, but also upon the likelihood and time required for successful restoration, as well as the direct impact of invasive species control measures on wildlife and their habitat (Bateman et al. 2008a).

The impact of invasive species on wildlife habitat, and therefore the benefits gained by restoring these habitats, may be relatively predictable based on what is known about the habitat requirements of particular species. For example, deer, elk, and bison rely heavily on grasses. When grasslands are invaded by weedy invasive forbs, grass production declines and animal use of these habitats can decline by up to 80% (Thompson 1996; Rice et al. 1997b; Duncan 2005).

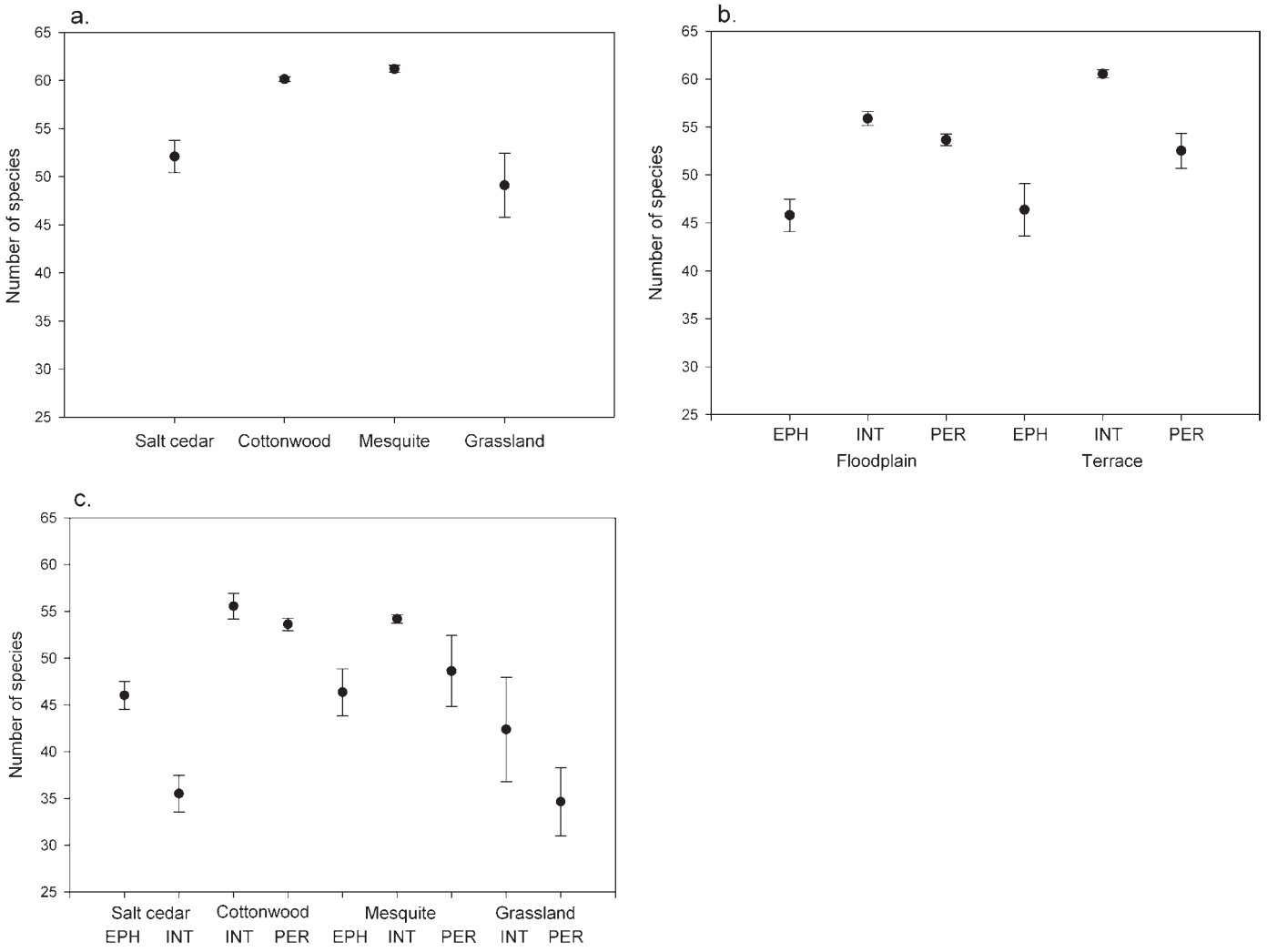


FIGURE 6. Avian species richness (mean ± SE) as a function of **a**, four vegetation types, **b**, three hydrologic regimes within riparian location (floodplain and terrace), and **c**, nine combined vegetation–hydrologic regime classes on the San Pedro River, Arizona, 1998–2001. EPH indicates ephemeral surface water flow; INT, intermittent surface water flow; and PER, perennial water flow.

Similar effects have been observed on bird populations that prefer open grasslands (Scheiman et al. 2003). In cases where invasive plants alter the preferred forage base or structural characteristics of the native plant community, restoring these systems likely will have a large positive effect on wildlife.

Even if community structure is not altered, restoring natural patterns of plant species abundance may greatly improve habitat. For example, in western Oregon grassland, tall oat grass (*Arrhenatherum elatius* [L.] P. Beauv. ex J. Presl & C. Presl) reduces grassland use and egg laying by the endangered Fender's blue butterfly (*Icaricia icarioides fenderi* Macy) (Severns 2008). Oat grass appears to reduce use largely by obscuring the butterfly's preferred host plant, Kincaid's lupine (*Lupinus oreganus* A. Heller var. *kincaidii* C.P. Sm.), even when the lupine is present for butterfly use. Plant invasions are likely to have the greatest influence on wildlife when their presence leads to feedbacks that not only change the plant community structure, but also alter ecosystem properties.

For example, in North American rangeland, cheatgrass probably has the most widespread and severe effects on wildlife of any invasive plant. By changing the fire regime, cheatgrass can displace shrub-steppe vegetation and

associated wildlife species. Furthermore, cheatgrass can lead to such changes over extremely large areas (Fig. 7). In the Intermountain West, it has been estimated to occupy 40 million ha (DiTomaso 2000). Of primary concern are rare species such as greater sage grouse (*Centrocercus urophasianus* Bonaparte), Gunnison sage grouse (*Centrocercus minimus*), Brewer's sparrows (*Spizella breweri*), sage sparrows (*Amphispiza belli*), sage thrashers (*Oreoscoptes montanus*), and pygmy rabbits (*Brachylagus idahoensis*). Sage grouse are considered to be sagebrush obligates (Schroeder et al. 2004), and are most likely to persist in large areas with at least 25% sagebrush cover (Aldridge et al. 2008). Both greater sage grouse and Gunnison sage grouse have been proposed for listing under the Endangered Species Act. Although prescribed fire has in the past been suggested as a tool to improve sage grouse habitat, recent studies suggest fire is most often harmful to sage grouse, particularly the frequent fire caused by cheatgrass invasion (Connelly et al. 2000; Baker 2006). Invasion of sagebrush by cheatgrass, and associated increases in fire frequency, appear to be primary causes of sage grouse decline (Knick et al. 2003; Schroeder et al. 2004; Baker 2006).

Other sagebrush-obligate wildlife species are likely to be similarly influenced by cheatgrass invasion and loss of sagebrush. Quantification of habitat requirements shows that many species considered to rely on sagebrush, such as pygmy rabbits, sage thrashers, and sage sparrows, do in fact have habitats that overlap strongly with those of sage grouse (Rowland et al. 2006). In eastern Washington, not only shrub-nesting sage sparrows, but also a variety of ground-nesting birds, were found to be less abundant in areas dominated by cheatgrass than in shrub–grass plant communities (Brandt and Rickard 1994). A variety of avian species have also been shown to prefer native perennial grass seed to cheatgrass seed (Goebel and Berry 1976). Small mammals can also be strongly influenced by cheatgrass invasion. For example bitterbrush-dominated communities have been found to support 3–13 times the densities of small mammals of cheatgrass-dominated communities in central Washington (Gano and Rickard 1982; Gitzen et al. 2001). Similarly, small-mammal densities and richness were higher in intact sagebrush steppe than in areas with cheatgrass in Idaho's Snake River Plain (Hanser and Huntly 2006). Finally, a study of Townsends ground squirrels (*Spermophilus townsendii idahoensis* Merriam), found higher variation in squirrel burrows in cheatgrass habitats than in shrub–bunchgrass habitats (despite similar mean burrow numbers), suggesting cheatgrass provides an adequate but unstable food resource for this species (Yensen et al. 1992).

Beneficial Effects of Invasive Plant Management on Wildlife. Only a handful of studies have actually measured effects of rangeland weed control on wildlife or wildlife habitat. Chemical control of spotted knapweed control was found to release grasses from competition and increase winter forage for elk by 47% in western Montana (Rice et al. 1997b). Other wildlife species rely heavily on native forbs for food. Consequently, restoration of native forbs can be an important objective of invasive species control. Fall wick application of glyphosate controlled Canada thistle (*Cirsium arvense* [L.] Scop.) while increasing shrub biomass, forb biomass, and species richness in a Montana waterfowl production area (Krueger-Mangold et al. 2002). Burning can also favor native forbs if conducted at the right time. In California, burning in the late spring and early summer

FIGURE 7. *Bromus tectorum* invasion in shortgrass steppe vegetation near Lander, Wyoming. (Photo: D. Blumenthal)

“There are too few direct measurements of wildlife responses to invasive species control to gauge how often control yields benefits for wildlife.”

after many forbs have set seed can favor native forbs over invasive annual grasses (DiTomaso et al. 2006b; Meyer and Schiffman 2008). Similarly, winter–spring cattle grazing appears to reduce annual grass invasion and favor native forbs that provide key habitat for several butterfly species on California serpentine grasslands (Weiss 1999b). A few studies have directly documented increased wildlife use following invasive species control. For example, successful control of spotted knapweed with picloram increased elk foraging in a Montana old field (Thompson 1996). Similarly, mechanical and chemical control of saltcedar and Russian olive (*Elaegnus angustifolia* L.) in New Mexico floodplain forest increased the abundance of lizards and bats, but decreased the abundance of birds that nest in midstory vegetation (Bateman et al. 2008a, 2008b).

Detrimental Effects of Invasive Species Management on Wildlife. Although reduced abundance of invasive species is likely to benefit wildlife, the methods used to reduce invasive species abundance can sometimes harm wildlife. A potentially important example of direct effects of herbicides on wildlife can be found in recent work suggesting that atrazine plays a role in global declines in amphibian populations (Rohr et al. 2008). Both field surveys in Minnesota wetlands and mesocosm experiments showed atrazine to be associated with increased infection by trematodes (a likely proximate cause of amphibian declines) in northern leopard frogs (*Rana pipiens* Schreber). Increased infection, in turn, appears to be caused by both an increase in the abundance of gastropods, which are intermediate hosts for trematodes, and decreased immune responses on the part of the frogs (Rohr et al. 2008).

Invasive species control can also harm wildlife indirectly, through its effects on the plant community. In particular, chemicals that target dicots can decrease plant community diversity, thereby reducing the food available for some wildlife species (Johnson et al. 1996a; Sheley et al. 2007). For example, 2,4-D applied to western Colorado rangeland to favor grasses over forbs and shrubs reduced densities of northern pocket gophers (*Thomomys talpoides* Richardson) and least chipmunks (*Eutamias minimus* Bachman), while increasing densities of montane voles (*Microtus montanus* Peale; Johnson and Hansen 1969). These effects appear to have been caused by a combination of reduced food availability and changes in vegetation cover. A proposed alternative for controlling woody species without harming wildlife is tebuthiuron, which can reduce woody species without reducing forb abundance and diversity (Johnson et al. 1996b).

Conclusions and Management Implications. There is considerable evidence that invasive plants influence wildlife in rangelands. Most often this influence is negative, reducing food and habitat availability for a wide array of wildlife species. Evidence for negative effects on wildlife is particularly strong for invaders that alter ecosystem structure and function, such as weedy forbs invading grasslands and annual grasses invading systems historically dominated by perennial plants. There are too few direct measurements of wildlife responses to invasive species control to gauge how often control yields benefits for wildlife. Rather, wildlife benefits must be inferred from what is known about both the relative value of invasive and native plant species as wildlife habitat, and the effectiveness of management in replacing invasive species with native species.

Protect Life and Property from Wildfire Hazards

Wildfires are a regular and natural occurrence in many areas of the arid western United States and most of these ecosystems are well adapted to fires (Brooks et al. 2004). These natural ecosystems will return to their preburn state within a few years of a fire under normal conditions. However, other habitats such as riparian corridors, sagebrush scrub, and deserts have longer fire-return intervals because of sparse and discontinuous vegetation. In these areas, the native species are less adapted to fire and are susceptible to a short-duration fire interval (Brooks et al. 2004). Invasion by annual grasses, particularly cheatgrass, red brome (*Bromus rubens* L.), and medusahead, have dramatically shortened the intervals between fires by providing more continuous fuels that are easier to ignite (Brooks et al. 2004). In addition, invasive annual grasses typically reestablish more rapidly than native plants after fires. This can further suppress the

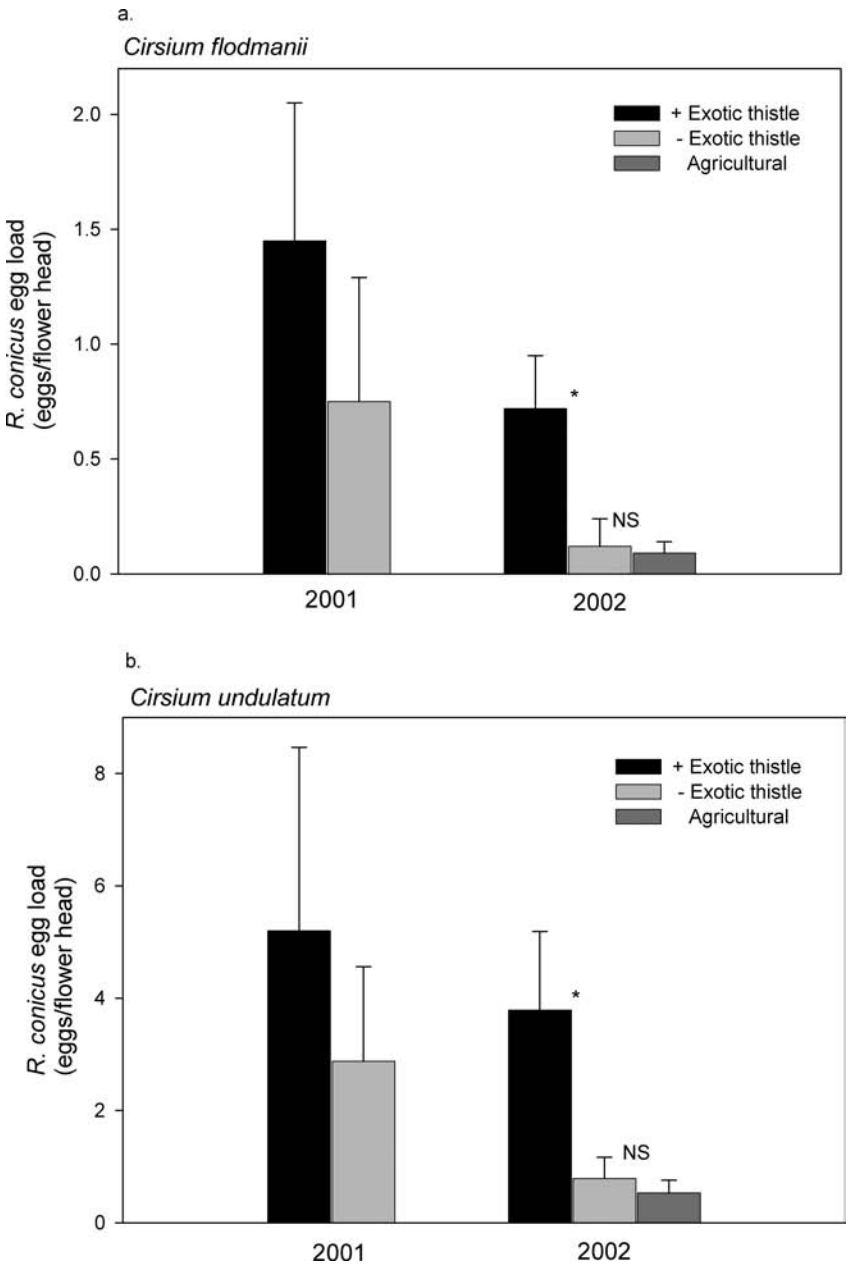
recovery of the natives and allow the weeds to expand their range (Pellant 1990).

More importantly, if fires occur too frequently, some of the native vegetation becomes so severely damaged that recovery is no longer possible (Pellant 1990; Whisenant 1990). This can result in loss of woody species such as sagebrush and other important plants and wildlife species, and effectively convert high-diversity native plant communities into low-diversity nonnative communities (Knick 1999). In some cases, fire exclusion over a period of time can create undesirable conditions for both forest sustainability and human fire hazard (Keeley 2006). This is the situation with some woody species, such as western juniper (*Juniperus occidentalis* Hook; Coultrap et al. 2008), which has expanded its range dramatically in the northwestern United States.

Land management agencies, such as the USDA Forest Service, Bureau of Land Management (BLM), and National Park Service, are required to assess site conditions following wildfire. Where necessary, they can prescribe emergency watershed-rehabilitation measures that can 1) help stabilize soil; 2) control water, sediment, and debris movement; 3) prevent permanent impairment of ecosystem structure and function; and 4) mitigate significant threats to human health, safety, life, property, or downstream values (USDA Forest Service 2010). Each year millions of dollars are spent on emergency post-fire rehabilitation treatments (Robichaud et al. 2000).

In southern California, where chaparral communities are prone to fire at the wildland–urban interface and the societal impacts of accelerated postfire erosion are enormous, there are pressures to treat burned hill slopes with grass seed to protect life and property (Gibbons 1995). It was common to seed such areas with quick growing annual plants, typically nonnative annual ryegrass or collections of native and nonnative forbs. This practice, however, is no longer recommended because the results are often unsuccessful. In some cases, heavy rains can wash away seeds, or inadequate rainfall prevents good seed germination. In addition, some of plants used for reseeding can persist and add to the invasive plant problem (Bell et al. 2007) by competing with the native vegetation and preventing recovery. Long-term slope stabilization is better achieved by promoting the recovery of deep-rooted perennial shrubs compared to shallow-rooted annuals. This can be accomplished by transplanting shrubs or by

FIGURE 8. *Rhinocyllus conicus* egg load (untransformed means ± 1 SE) on two native thistle species—**a**, *Cirsium flodmanii* and **b**, *Cirsium undulate*—in grassland patches within two landscape types in 2001, and three landscape types in 2002. Results of planned contrasts comparing landscape pairs are presented above bars for 2002. NS indicates not significant; $P < 0.05$.



protecting establishing shrubs from herbivory or competition from nonnative species. Shrub recovery can lead to reduced threat of subsequent fires (Bell et al. 2007).

Conclusions and Management Implications. There is a considerable amount of evidence to demonstrate the impact of invasive plants, particularly annual grasses, on the frequency of fires in rangeland systems. In addition, it is well recognized that rangeland fires spread by invasive plants can cause significant damage to property and human health. Although few studies have been conducted on the interaction between invasive plants, wildfire, and impacts to wildlife, it stands to reason that these impacts are significant and in most cases detrimental to wildlife. With increased research on methods to control vegetation and protect areas from large catastrophic fires, the economic and ecological damage caused by invasive plants can be substantially reduced in the future.

Minimize Negative Impacts of Pest Control on Soil, Water, Air, Plant, and Animal Resources

Minimizing negative impacts of pest control on biotic and abiotic resources is an important step in designing economically and ecologically sustainable invasive plant management practices (Sheley et al. 2010). The most commonly applied control strategies for invasive plants on rangeland include herbicides, biocontrol, grazing, fire, or mechanical control such as tilling (Jacobs et al. 1999). Impacts of these control strategies on abiotic and biotic resources have been assessed to varying degrees and in some cases, general ecological patterns and principles are beginning to emerge. For example, the fate and ecological impact of various herbicides on rangelands has been documented and there is much evidence suggesting that as disturbance (e.g., herbicide use, tilling, grazing) intensity increases, invasibility of a system also increases (Hobbs and Huenneke 1992; Davis et al. 2000). Nevertheless, large gaps in our understanding of pest control impacts on abiotic and biotic resources remain. For example, a key component of ecologically based invasive plant management is to apply pest control strategies that reduce the performance of invasive species more than the performance of desirable species (Sheley et al. 2006). However, a Web of Science query that included the search terms "herbicide" and "rangeland" demonstrated that only 28% (20 of 70) of field studies published between 1976 and 2008 examined herbicide effects on both desirable and weedy vegetation.

Impacts of Control on Soil, Water, and Air Resources. Impacts of pest control on soil, water, and air resources vary depending on pest control strategy, but in general, effects are relatively predictable. For example, intense soil disturbances contribute to erosion, decreased water quality, and dust production and also release nutrients, which favors the growth of weeds compared to natives (Greene et al. 1994; Davis et al. 2000; McEldowney et al. 2002; Zhao et al. 2005). Because of this, current management frameworks for weed-infested rangeland focus on using tools that will minimize disturbance such as no-till drills and moderate grazing in efforts to direct a plant community toward a more desirable state (Mangold et al. 2006).

Concerns over the effects of herbicides on soil, water, and air resources have been raised due to the potential of herbicides to affect soil processes, to contaminate groundwater, or to be transported on wind-eroded sediment and potentially inhaled by humans (Larney et al. 1999; Liphadzi et al. 2005; Borggaard and Gimsing 2008). The impact of these herbicides on these resources is dependent on type of herbicide used, application rate, and soil characteristics, among other factors. For example, glyphosate tightly adheres to soil, which makes it difficult for this compound to leach into groundwater or affect soil biological processes (Borggaard and Gimsing 2008). On the other hand, compounds such as dicamba and picloram are highly mobile in the soil (Krzyszowska et al. 1994). High application rates, high rainfall following application, or direct application of these compounds to water bodies can pose a significant threat to water resources. Overall, careful application of herbicide following recommended procedures coupled with the relatively low application rate of herbicides commonly used on rangeland tends to minimize the negative effects of herbicides on rangeland soil, water, and air resources.

Impacts of Pest Control on Plant and Animal Resources

Herbicides. Only a subset of studies (20 of 70) has examined herbicide effects on both invasive and desirable plant species in the field. Although the responses are dependent on a number of factors, such as mode of herbicide action and site-specific environmental conditions, two important trends have emerged. First, desirable species functionally or taxonomically similar to the invasive plant species targeted for control tend to be more negatively impacted by herbicide application. For example, herbicides such as 2,4-D, clopyralid, or picloram are commonly applied to control broadleaf weeds such as knapweed, leafy spurge, and sulfur cinquefoil on rangeland. Because grasses are capable of metabolizing these compounds, desirable rangeland grasses are generally unaffected by these herbicides (Sheley and Jacobs 1997; Sheley et al. 2002; Laufenberg et al. 2005). However, these herbicides can greatly decrease native forb density and cover (Sheley and Denny 2006). There is evidence suggesting that herbicide effects on native forbs are long-lasting and can drive a local decline in species richness (Fuhlendorf et al. 2002; Rinella et al. 2009). As another example, desirable rangeland grasses have shown varying degrees of susceptibility to imazapic, a herbicide used to control invasive annual grasses, with evidence suggesting grasses within the Hordeae tribe may be more tolerant to imazapic than other grass species (Kyser et al. 2007). A second trend is that the impact of herbicides on desirable vegetation depends on the rate and timing of herbicide application. In general, when herbicides are applied several weeks prior to seeding or during a dormant seeding, herbicides have a greater selectively for weeds compared to seeded species (Jacobs et al. 1999; Kyser et al. 2007; Sheley 2007). Higher herbicide application rates can have negative impacts on seeded species, even during fall dormant plantings (Monaco et al. 2005). Even with a given rate and timing of herbicide application, desirable species response can vary substantially across sites in a given year and across years in a given site (Monaco et al. 2005; Sheley et al. 2007). Beyond a few generalities, the effect of herbicide on desirable vegetation remains difficult to predict.

The impacts of herbicides on desirable and undesirable species depend on the rate and timing of the herbicide application. (Photo: Rob Wilson)

Effects of herbicides on mammals, birds, and invertebrates are generally identified during the ecological risk assessment prepared with each herbicide and for public land management activities during environmental impact reporting. Although a number of herbicides are available to control weeds on rangeland, 70% of the land treated with herbicides by the BLM uses 2,4-D, glyphosate, picloram, tebuthiuron, or imazapic. Of these, glyphosate, picloram, and imazapic show low toxicity to terrestrial animals whereas tebuthiuron and 2,4-D demonstrate moderate toxicity. The low rates of herbicide applied on rangeland combined with relatively low toxicity and lack of chronic exposure suggest herbicides have minimal effect on terrestrial animal species on rangeland.

Biocontrol. Development and release of biocontrols follows international and national guidelines designed to minimize the possibility that biocontrol releases will negatively impact desirable vegetation (FAO 1996; Wilson and McCaffrey 1999). Biological control has been implemented successfully in a number of systems (e.g., Huffaker and Kennett 1959; McEvoy et al. 1991; Lym 2005) and when operating under current protocols there are relatively few documented direct effects of biological control on desirable vegetation given the number of biocontrol releases

(Funasaki et al. 1988; Center 1995). There is, however, mounting evidence suggesting that poor monitoring efforts, difficulty in predicting biocontrol effects, and the largely unrecognized indirect effects biocontrols can have on ecosystems contributes to an underestimation of the detrimental effects of biocontrols on desirable vegetation (Simberloff and Stiling 1996; Thomas and Willis 1998; Pearson and Callaway 2008). For example, the bulk of biocontrol monitoring focuses on release sites with little attention paid to offsite biocontrol effects even though there is strong evidence demonstrating landscape-scale variation in biocontrol effects on desirable vegetation (Simberloff and Stiling 1996; Rand and Louda 2004). In addition, it is estimated that less than half of the biological control efforts targeting invasive plants in the United States demonstrated any evidence of control (OTA 1995). Given that our ability to predict biocontrol effects on well-studied target vegetation is so low, some researchers have questioned the ability to predict biocontrol effects on desirable vegetation (Thomas and Willis 1998). Although examples of direct effects of biological control on desirable vegetation in a number of systems supports these concerns (Simberloff 1992), of equal importance is the recent literature showing complex indirect effects of biocontrol on desirable vegetation. For example, following the collapse of the target pest population, intense competition among biocontrol agents can cause a transient increase in host plant range, which results in the biocontrol agents attacking desirable vegetation (Lynch et al. 2002). Alternatively, when biocontrol agents only moderately damage invasive plants they may increase invasive plant competitive ability by stimulating compensatory growth (Callaway et al. 1999). In this situation, moderately damaged invasive plants may serve to maintain biocontrol densities at high levels, increasing biocontrol impacts on desirable vegetation (Rand and Louda 2004; Figs. 8a and 8b). These patterns of responses suggest, at a minimum, that current procedures do not adequately prevent biocontrol efforts from having significant impacts on desirable vegetation.

Concerns over the effects of biocontrol on animal resources largely have been centered on within-guild (e.g., insect) interactions. For example, if an introduced biocontrol insect shares food sources or parasites with a native insect, then biocontrol can have direct and indirect effects on native insect populations (Louda et al. 1997; Willis and Memmott 2005). An analysis of 17 food webs in Australia showed that a weed biocontrol agent with high weed host specificity was associated with a decline in native insect diversity (Carvalheiro et al. 2008). Although the magnitude of these direct and indirect effects are difficult to quantify and are generally underreported in the literature, basic community ecology theory predicts that such affects may be common (Holt 1977). In some cases, however, effects of biocontrol on a desirable plant community can be complex and difficult to predict, involving multiple interactions within a food chain. For example, introduction of gall flies to control spotted knapweed dramatically increased deer mouse populations that used gall flies as a food source (Ortega et al. 2004). Because deer mice also use native plant seed as a food source, introducing gall flies increased deer mouse populations which resulted in increased predation on native seeds and overall decrease in native plant density (Pearson and Callaway 2008). Although theory and empirical evidence suggest biocontrols likely will have some negative effect on native animal populations, biocontrol may still be an appropriate option if benefits outweigh the costs. Namely, if biocontrols have a large negative effect on weed populations, this benefit may outweigh moderate negative impacts of biocontrol on native plant and animal populations.

Grazing. Prescribed grazing effects on nontarget vegetation depend on a number of factors, including animal species used, timing of grazing relative to the phenology of desirable vegetation, and forage quality and quantity of weedy vegetation relative to desirable vegetation, as well as grazing tolerance of weedy and desirable species. Moderate grazing using animals or mixtures of animals (e.g., sheep and cattle) that demonstrate certain dietary preferences for a particular weed can be used to decrease weed density and increase density of desirable plants (Bowns and Bagley 1986; Sheley et al. 1998). In general, when grazing is limited to periods when weedy species are most susceptible to defoliation and desirable plants are largely dormant, the impact of grazing on

Prescribed grazing can be an effective tool in reducing the vegetative growth of invasive species. (Photo: Brenda Smith)

desirable vegetation can be minimized and benefit of grazing for weed control maximized (Kennett et al. 1992). For example, utilization of grasses by sheep in areas infested with knapweed was decreased by timing grazing to occur when knapweed was still growing and vegetative growth of desirable grasses had largely stopped for the season (Thrift et al. 2008). If weedy and desirable vegetation have comparable forage quality, grazing animals largely will consume plants in proportion to their abundance. For example, diets of sheep used to graze spotted knapweed were over 50% grasses in areas with low spotted knapweed density, but were less than 20% grasses in areas with high spotted knapweed density (Thrift et al. 2008). When weeds have much lower forage quality compared to desirable vegetation, grazing animals can have a larger preference for and a much greater negative impact on the desired vegetation (Ralphs et al. 2007). Despite these general guidelines it is difficult to predict the effect of grazing on desirable vegetation. For example, grazing leafy spurge infestations has been found to decrease (Jacobs et al. 2006), increase (Seefeldt et al. 2007), or have no effect on (Lacey and Sheley 1996) the cover of desirable grasses. Although a portion of this variation may be explained by differences in grazing systems, differences in grazing tolerance between weedy and desirable species at a particular site also may be important (Kennett et al. 1992; Kirby et al. 1997; Olson and Wallander 1997). If weedy species demonstrate a greater tolerance to grazing than desirable species do, then prescribed grazing may be a counterproductive control strategy even if grazing animals demonstrate greater or equal preference for weedy species compared to desirable species (Kimball and Schiffman 2003). Although the value of prescribed grazing for weed control has been demonstrated in a number of systems, negative impacts on desirable vegetation have been demonstrated, highlighting the need to closely monitor prescribed grazing efforts.

Conclusions and Management Implications.

A key step in designing economically and ecologically sustainable invasive plant management practices is to apply management techniques that minimize negative impacts

on biotic and abiotic resources. Some general principles are beginning to emerge allowing progress to be made toward this goal, such as our understanding of the relationship between disturbance intensity and invasibility. Although some negative effects of pest control strategies on native plant and animal resources are likely, herbicides, biocontrol, and grazing can be applied in ways that greatly minimize these impacts if ecological processes and mechanisms are considered beforehand and control strategies are adjusted to address these factors. Identifying these processes and mechanisms and making necessary adjustments in management, however, is far from straight forward. Complex direct and indirect effects of control efforts on desirable plant and animal resources occur, requiring careful implementation of control efforts, comprehensive monitoring, and a broad determination of costs and benefits achieved by control efforts that include multiple ecosystem components.

RECOMMENDATIONS AND KNOWLEDGE GAPS

Our recommendations are centered on three general aspects of invasive plant management. The first revolves around improving and standardizing data collection and risk analysis needed to better inform management decisions. Second, progress toward science-based management of rangeland threatened and/or dominated by invasive species must be greatly accelerated. Third, invasive plant management would greatly benefit from the development and implementation of a comprehensive education and technology transfer program. The objective of this portion of the document is to provide critical recommendations to guide future development of invasive weed management and to identify important knowledge gaps. A brief rationale and justification for each recommendation and knowledge gap are provided as well.

Standardized Data Collection, Risk Analysis, and Prioritization Procedures

The magnitude and complexity of invasive plant management requires that ecologists garner maximum information from all datasets. Data collection for both invasive plants and desired species is central to developing appropriate management programs in the future. Standardized data collection will be required in order to allow data comparisons among years and data combinations to conduct meta-analysis needed for development of robust principles for management. Managers need standardized data collection procedures to create accurate vegetation assessments that allow periodic evaluations of their management. Inventory data must be summarized and analyzed to forecast likely future vegetation patterns so ecological and economic risk/benefit analysis can be accurately conducted. Standardized ecological and economic data collection would be critically valuable to determine land areas with characteristics that favor the likelihood of success in response to a particular control strategy.

Science-Based Solutions to Invasive Plant Management

Just as physics provides the scientific principles for engineering, ecology must provide the scientific principles for invasive plant management. We strongly recommend further development of ecologically based management frameworks that can be used to guide the incorporation and application of ecological principles for invasive plant management. Frameworks must be useful to researchers and managers, so the connection between these complementary endeavors is natural and direct. State-and-transition models that utilize ecological processes and the influence of management on these processes to predict vegetation dynamics represent a viable framework for various ecological site descriptions. A process- and evidence-based approach is central to advancing invasive plant management from misapplied treatments that address only symptoms to management programs that emphasize the underlying cause of invasion, retrogression, and succession.

Complex interrelationships among various components within ecosystems create multiple indirect responses to vegetation management that are very difficult to predict. This creates a strong need to manage invasive plants within the context of the entire ecosystem. Invasive plant management must become more integrated within a systems approach to

Invasive species management that addresses the actual cause of invasion and employs ecological principles in management strategies increases the likelihood of success. (Photo: Chris Call)

facilitate problem solving and the attainment of well-defined goals, rather than only practice-based outcomes. Management must assess the complex interrelationship among ecosystem components and processes and design management strategies that influence the underlying ecological cause of invasion and dominance by invaders with predictable outcomes.

Imposing management that addresses the actual cause of invasion is clear in some cases. For example, the increase in invasive wetland species in flooded waterfowl habitat on the Malheur National Wildlife Refuge requires flooding regimes to be less frequent, allowing substantial dry periods to shift the balance in favor of diverse vegetation. In many cases, the actual causes of invasion are less obvious and may actually be a result of multiple direct and indirect interactions that determine successional dynamics. Thus, weed ecologists and scientists must develop guidelines to evaluate causes of invasion, succession, and retrogression. Once these guidelines are developed, ecological principles must be developed that provide guidelines for managers to impose tools and strategies to influence conditions, mechanisms, and processes in favor of desired vegetation. As multiple interactive ecological processes require amendment, integrated plant management strategies can be developed and employed much more effectively. In this way, various plant management strategies can be designed based on how the treatments influence the ecological processes that direct ecosystem change. Tools and strategies that are based on sound ecological principles could enhance our ability to employ effective integrated management.

Enhancing our ability to prevent invasion is critical for successful implementation of integrated invasive plant management. Given the complexity and persistence of invasive plants, a proactive approach focused on systematic prevention and early intervention

Practicing prevention of invasive species, such as medusahead, is more economical and more effective than costly restoration. (Photo: Ryan Steineckert)

could be much more effective than the existing reactive approach. Most managers recognize the importance of prevention, but lack the ability to effectively employ it. Science-based prevention strategies that are based on the ecology of seed dispersal are severely needed. Land managers need a conceptual framework and associated tools that assist them in indentifying which vectors are major contributors to invasive species dispersal and propose dispersal management strategies to minimize or interrupt these major vectors. Effective methods for containing existing infestations are also needed.

Invasive plant management is currently applied in a somewhat haphazard way based on political pressure and funding resources. In the future, more emphasis should be focused on prioritizing invasive plant management in areas that have the highest likelihood of success both economically and ecologically. Methods for prioritizing invasive plant management will continue to be increasingly necessary as a means to effectively allocate scarce resources. Moreover, the lack of successful control of invasive species indicates that we may transition toward a management philosophy that minimizes the negative impacts of invasive species and maximizes the ecological and economic benefits garnered from invasive weed management programs. Concepts, such as economic/ecological injury levels, biomass optimization models, and thresholds will need to be carefully developed in a manner that helps managers prioritize management programs to address invasive species.

Comprehensive Education and Technology Transfer Programs

Although some of the necessary infrastructure to conduct educational programs effectively is in place, a unified, progressive, and outcome-based educational and technology transfer program would have strong synergistic effects on invasive plant management. Educational programs vary widely in their objectives, content, and outcomes. Current programs lack continuity of message and the ability to progressively advance managers' understanding of science-based management. We propose that various ecological societies, managers, and researchers develop a comprehensive science-based curriculum promoting the most state-of-the-art, science-based assessment and management strategies. Once developed, training modules could be developed for various portions of the educational infrastructure having responsibility for natural resource extension and outreach.

Restoration of invasive plant–dominated rangeland is extraordinarily risky and expensive. Based on our assessment, the continued application of "farming system" seeding methods is unlikely to provide sustainable replacement of invasive species. Many ecological barriers to seed germination, seedling establishment, and population development exist in restoration areas where

invasive plants dominate. Managers must have an understanding of these barriers and methods for overcoming them if restoration is to become a useful strategy to restore previously invaded sites and prevent reinvasion in the future. Restoration approaches must be founded upon ecological principles that can be applied to specific sites and varied as environmental conditions vary across landscapes.

Invasive plant problems and solutions are complex and management outcomes are rarely predictable. Ecologists and managers are often uncertain about the best management practices to employ, or if management will actually repair plant communities and the associated ecological processes. In most cases, simple answers to complex situations do not exist and solutions to invasive plant problems are elusive. Managers need scientifically credible methods for testing various management strategies that can be used when management programs are being planned and implemented. These adaptive management strategies should include controls for comparisons and designs that use simple experimental hypothesis testing, in addition to monitoring previous effectiveness. A major strength of adaptive management is that it would allow managers to continuously evaluate the effectiveness of current invasive plant management programs and assist with identification of the most successful management programs.

CONCLUSIONS

Invasive plants negatively impact rangelands throughout the western United States by displacing desirable species, altering ecological processes, reducing wildlife habitat, degrading systems, altering fire regimes, and decreasing forage productivity. Assessing the influence of conservation practices on the perceived benefits to ecosystems is critical to understanding their usefulness in maintaining sustainable ecological and economic systems. We conducted a comprehensive synthesis of peer-reviewed literature to determine the efficacy of various invasive plant strategies on several anticipated benefits. The literature documented only short-term vegetation responses to invasive plant management and rarely addressed long-term ecological outcomes associated with invasive plant management. Our ability to protect noninfested lands is encumbered by the lack of early detection techniques and effective eradication efforts once new infestations are identified. Several strategies for maintaining invasion-resistant plant communities are beginning to emerge. Herbicides provided short-term control of most invasive weeds, but without additional management, weeds often return rapidly. Documentation of the efficacy of biological control on plant development is well established, but positive effects on control and vegetation dynamics are exceedingly rare. Grazing management is emerging as a useful method for managing invasive plant species, but the timing, intensity, and frequency of grazing, as well as the class of livestock are only known for a few invasive species. Restoration of infested rangeland is difficult and only successful about 20% of the time when nonnative plant material is seeded and the probability is even less when native species are used. There are cases in which invasive plant management strategies can be effective, and in those cases, the management strategies appear to favorably affect wildlife and other important ecological attributes of ecosystems. However, most strategies are associated with high ecological risks and high risk of failure in the long term. It is clear that more research is necessary if the anticipated benefits of invasive plant management are to be achieved. This synthesis indicates that long-term invasive plant management is lacking for most applications and that ecologically based invasive plant management is desperately needed to meet this escalating problem on rangelands.

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As invasive plants continue to spread across the landscape, a priority should be developing comprehensive education and technology transfer programs. (Photo: Ryan Steineckert)

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CHAPTER

8

A Landscape Approach to Rangeland Conservation Practices

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> “…field-scale (local) evaluations of conservation practices that have been emphasized in this document do not enable evaluation of the *broader-scale, cumulative effects of practices*”

A Landscape Approach to Rangeland Conservation Practices

8

Brandon T. Bestelmeyer, Joel R. Brown, Sam D. Fuhlendorf, Gene A. Fults, and X. Ben Wu

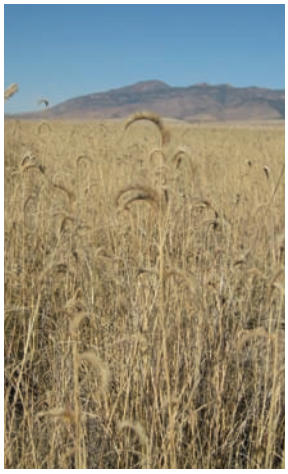
INTRODUCTION

A primary objective of the Conservation Effects Assessment Project (CEAP) has been to evaluate whether or not rangeland conservation practices (hereafter "practices") supported by the US Department of Agriculture Natural Resources Conservation Service (USDA NRCS) yield the environmental benefits that we ascribe to them. In doing so, the authors of most CEAP chapters have focused on specific conservation practices (e.g., grazing management or brush control) and sought generalities about their effectiveness based on a review of the literature. Often, a weight-of-evidence–based interpretation is drawn from the percentage of studies that support a particular assertion. The goal of this approach is to produce general recommendations about the implementation of practices, sometimes tailored to broad ecosystem types. The limitations of this approach, however, are revealed when the weight of evidence for or against the utility of a practice is equivocal. Often, practices both succeed and fail in different situations. Thus, the evidence suggests that we do not have the data to discover *under what circumstances a practice succeeds or fails* (Michener 1997). In other words, *we cannot yet account for spatial heterogeneity*, including variations in soils, climate, vegetation state, within-site patchiness, and landscape position relative to dispersal, water movement, and other processes that strongly influence the success of practices. In the face of this information shortage, we inevitably overgeneralize and fail to recognize important variations in context.

A related issue is that many practices are believed to have benefits that are manifested at spatial scales larger than the treatments themselves. A primary example is the expectation that prescribed grazing or brush management applied to uplands will have measurable effects on riparian or watershed function (Goodwin et al. 1997). It is also expected that practices will have a measurable positive impact on rangeland conditions at a landscape to regional level and that an improvement in one location is not offset by degrading processes at another location. Thus, field-scale (local) evaluations of practices that have been emphasized in this document do not enable evaluation of the *broader-scale, cumulative effects of practices* (e.g., Kondolf et al. 2008). Because of both spatial heterogeneity and differences in how multiple local treatments scale up to affect broad-scale attributes, we cannot simply assume that more is better in a linear way. We will have to measure directly attributes at broad scales and relate them to the locations and consequences of field-scale practices. Such linkages are currently rare because conceptual models of cross-scale interactions are only now being developed and resource managers are usually not certain how to apply them.

Similarly, from a sociological standpoint, there is an expectation that successful practices accelerate their adoption by other landowners in the immediate area (Kreuter et al. 2005). Among the hypothesized benefits of conservation programs, including technical assistance and cost sharing, is that local demonstration of benefits encourages the use of practices among neighboring managers. The spread of practices among managers provides another means for local practices to have effects at broader scales.

In this synthesis, we promote the development of a systematic approach by which the NRCS



Blue grama (Bouteloua gracilis) grass (foreground) and the Animas Mountains (background) in the Malpai Borderlands of southwestern New Mexico. (Photo: Brandon Bestelmeyer)

“…information provided by this “landscape perspective” could enable planners to increase successful application, use federal resources more efficiently, and assess more effectively the consequences of practices.”

and other agencies can evaluate both the local and *landscape context* of practices—that is, where they occur in a landscape and region and the varying processes and constraints associated with those locations. We further emphasize that this approach should include increased attention to *spatial pattern* as an attribute that contains valuable information, in addition to averages or sums of variables (such as plant cover) that are typically emphasized. Collectively, the information provided by this “landscape perspective” could enable planners to increase successful application, use federal resources more efficiently, and assess more effectively the consequences of practices. The empirical basis for these assertions within the rangeland conservation literature is weaker than for other chapters due to limited development of landscape perspectives in rangeland ecology and the consequent paucity of studies. Nonetheless, evidence from the broader literature in landscape ecology (e.g., Liu and Taylor 2002) and some key examples in rangelands supports our contention that it is essential for NRCS and its partners to develop 1) interpretive tools that facilitate a consideration of landscape context and spatial pattern in conservation planning and assessment and 2) database systems that link practice effects to ecological sites, state-and-transition models (STMs), and the mosaic of ecological sites and states in a landscape.

This chapter is organized into seven sections. Following this introduction, we review the current processes used by NRCS in conservation planning at different spatial scales. We then review concepts that can be used to better place practices in a landscape context and introduce a spatial hierarchy to facilitate application of landscape concepts. We then outline a model-based approach that could be used to design and test the effects of practices, taking into account landscape context and linking tests to ecological site descriptions (ESDs) and STMs. We offer some general recommendations for incorporating landscape perspectives in conservation planning, identify knowledge gaps that must be overcome to act on some recommendations, and conclude that landscape perspectives are useful and feasible. Because the language used to describe elements of the landscape perspective is not well-known or standardized, we encourage readers to refer to definitions for terms and phrases used in this chapter (Table 1).

CURRENT STATE OF LANDSCAPE PERSPECTIVES IN CONSERVATION PRACTICES

Conservation Planning at Multiple Scales

Most NRCS staff currently involved in conservation planning and implementation have been trained as “progressive” planners, which means that NRCS planners and clients incrementally implement conservation practices within a specified area. Progressive planning has the advantage of focusing resources on immediate concerns, but lacks the spatial and temporal perspectives necessary to meet landscape-scale goals and, more importantly here, to provide a consistent and transparent basis to assess conservation effects. The end result of the planning process should be a conservation plan that identifies specific actions to be taken by land managers in order to meet objectives for specific land areas. For a variety of reasons, progressive planning often does not result in a comprehensive strategy that addresses the variety of conservation needs at different scales.

Regardless of the planning approach, conservation plans rely on the implementation of individual or combinations of conservation practices to achieve objectives. “Conservation practices” are protocols for actions taken by land managers to improve or maintain the condition of rangelands (USDA NRCS 2003a). Practices are classified as 1) vegetation management practices, 2) facilitating practices, or 3) accelerating practices. To a large extent, this classification also reflects the amount of resources required to implement the practices:

1. Vegetation management practices are intended to influence the use and growth of the vegetation and are specifically evaluated in other chapters. Examples include prescribed grazing and prescribed burning.
2. Facilitating practices are intended to create infrastructure that aids in vegetation management and are only indirectly evaluated in other chapters. Examples

TABLE 1. Glossary of landscape-related terms and phrases used in this chapter. Many definitions adapted from Turner et al. (2001)

Cross-scale interactions	How processes at one spatial or temporal scale interact with processes at other scales (e.g., fine-scale plant growth interacting with flows of surface water in a landscape)
Land unit/plant community	Areas of land that are sufficiently large to be of management interest and are ecologically homogeneous with respect to management issues
Landscape	An area that is spatially heterogeneous in a property of interest, usually with respect to plant communities
Landscape/spatial context	The influence of the location within a landscape or in space, including both underlying heterogeneity and spatial interactions with neighboring locations
Landscape level/scale	A level of biological organization characterized by a mosaic of plant communities that have developed in response to common soil and geomorphic processes and that often exhibit spatial interactions (e.g., a watershed or multiple watersheds)
Multiple scale	Simultaneous consideration of patterns and processes occurring at different spatial scales
Patch	A relatively homogeneous area that differs from its surroundings; used here in reference to distinct fine-scale assemblages of plants or ground cover that make up a plant community
Patchy/Patchiness	Being made up of different patches; the degree to which an area is made up of diverse patches
Scale	Spatial dimension of a measured attribute or process, characterized by its grain (smallest resolved unit) and extent (the area across which measurements are taken)
Scale of heterogeneity/spatial pattern	The idea that heterogeneity and pattern can be defined differently at different spatial scales
Scaling up	Using measurements of properties gathered at finer scales to estimate properties at broader scales
Spatial heterogeneity	Variability or dissimilarity of properties of interest across a defined area
Spatial interaction	The flow of matter, disturbance, or information from one location to another
Spatial or landscape pattern	The arrangement of patches or land units of interest in geographical space and relationships between these units

include water developments, stock trails, and fencing.

3. Accelerating practices are intended to supplement vegetation management by promoting plant community change more rapidly than is possible through vegetation management alone, often at great expense. Examples are brush management, range planting, or channel stabilization.

Conservation planners work with individuals and groups to inventory resources, identify concerns and objectives, and develop a conservation plan. Once all of the planning inputs have been documented, individual practices are assembled into conservation systems to meet the needs of clients (e.g., a resource management system). This process involves map-based decisions about where to implement particular practices within individual field and property boundaries. Although this level of planning may involve a high degree of precision regarding the placement of practices (i.e., fencing, water development, roads), there is seldom a clearly defined link to important processes occurring at larger spatial scales. For example, a water development may be planned and implemented within an individual pasture with the objective of improving grazing distribution. However, the improved grazing distribution is seldom documented quantitatively and the assumed larger-scale effects of improved grazing distribution (water quality, habitat improvement) also lack consistent measurement. Notable exceptions to this generalization include cases in which planning considered sage grouse movements between nesting and lekking sites (Connelly et al. 2000) or where planning was designed to prevent

A semiarid grassland (foreground) and creosotebush shrub savanna (background on hill) in southcentral New Mexico. (Photo: Brandon Bestelmeyer)

the spread of waterborne bacteria (*Escherichia coli* O157:H7) from rangelands into green leafy produce fields (Tian et al. 2002; Jay et al. 2007).

One limitation to the broader use of landscape perspectives is the current focus on the relatively homogeneous "site" as the fundamental spatial unit of rangeland inventory, planning, and assessment (Brown et al. 2002; Washington-Allen 2006). The size of a site varies, but it is roughly 5–50 ha. Following current NRCS protocols, sites are classified to an "ecological site" based on the relationship of potential vegetation to differences in climate, soils, and landscape position relative to water movement or solar energy inputs. The potential vegetation and associated vegetation dynamics described for ecological sites (using STMs) have been used as a benchmark with which to gauge the effects of management on rangelands. The current ecological site system defines benchmarks on a "piece-by-piece" basis, however, and does not directly address how the composition and arrangement of multiple sites (i.e., landscape context) or variability within sites should influence decisions (USDA NRCS 2003a). Research in landscape ecology and the practical experiences of planners suggest that the landscape context of a land unit can be critically important (explanation below). Although management units commonly contain multiple ecological sites, there has been little attempt to formalize procedures for using information about the composition and arrangement of sites or variability within sites in planning or assessment. Historically, planning and implementation of practices did not consider

landscape context in the placement of water facilities, fences, or other facilitating practices. Multiple-site planning most often occurred when designing grazing systems to accommodate livestock movement, unique plant growth patterns, and nutritional needs. However, there has been little effort to search for principles relating to the assessment of practice effects at multiple spatial scales. Furthermore, there is little guidance for integrating multiple properties into planning or assessment, which represents an even broader scale of heterogeneity and pattern.

There is some precedent for planning and assessment at multiple scales in the form of "area-wide conservation plans" referred to in the *National Planning Procedures Handbook* (USDA NRCS 2003b).

"Area-wide conservation plans are voluntary, comprehensive plans for a watershed or other large geographic area. Area-wide conservation plan development considers all natural resources in the planning area as well as social and economic considerations. Plan development follows the established planning process to assist local people, through a voluntary locally led effort, to assess their natural resource conditions and needs; set goals; identify programs and other resources to solve those needs; develop proposals and recommendations to do so; implement solutions; and measure their success."

This statement allows for the scale of the Conservation Management Unit to be determined by the planner and acknowledges that goals can be defined for multiple scales, but provides little guidance for planning and assessment across the scales.

The Watershed Protection and Flood Prevention Act (83-566), referred to as PL 566, authorizes NRCS to work with clients at spatial scales greater than the property level. Guidance for this program is found primarily in the *National Watershed Manual* (USDA NRCS 2009). The tools for planning, however, are primarily those used for individual property planning. Assessment tools differ slightly in that outcomes are usually expressed at the watershed scale, often using economic variables (e.g., flood prevention benefits).

The *National Biology Handbook* (USDA NRCS 2004) also considers multiple scales for programs to enhance wildlife habitat. In this document, the concepts of core reserves (nodes), corridors, and buffer zones are introduced and applied by example. In addition, qualitative metrics for assessing patch, corridor, matrix, and structural attributes are introduced. In discussions of these attributes, it is clearly stated that planning, implementation, and assessment may span several spatial scales.

In spite of these precedents, NRCS policy, guidance documents, and reports of accomplishments have focused largely on planning and outcomes at the individual management unit or ranch scale. In addition, the dominance of the Environmental Quality Incentives Program (EQIP) in agency activities often dictates that planners work on Conservation Management Units that are defined by management unit boundaries. The lack of tools to assist in planning or quantitative assessment of ranch-, watershed-, or landscape-level outcomes limits the ability of planners to address broad-scale problems.

Measurement of Practice Effects

There is currently no specific NRCS protocol or program designed to measure conservation effects at multiple spatial scales. The only dataset that covers all nonfederal rangelands (and all other nonfederal lands) is the National Resource Inventory (NRI). The NRI is a "longitudinal survey of soil, water, and related environmental resources designed to assess conditions and trends every five years on non-federal US lands" (Nusser and Goebel 1997). The sampling framework for NRI is based on sampling areas (primary sampling units or segments) of 160 acres (64.8 ha). The number of segments has varied throughout the life of the NRI, ranging from 108 000 to over 300 000 nationwide. Within each segment, three sample points are randomly located and various attributes are measured in them. This structure permits statistical inferences about changes in land use and practice application at regional, state, and national levels. The NRI was designed initially to detect changes in broad classes of land cover and use (e.g., cropland to pasture, or wheat to corn). Most NRI analyses are based on photointerpretation. In 2003, a special study was established for

"There is currently no specific NRCS protocol or program designed to measure conservation effects at multiple spatial scales."

TABLE 2. Examples of specific applications of landscape concepts discussed in this chapter to rangeland conservation practices discussed in other chapters.

Conservation practice type	Spatial heterogeneity	Spatial pattern	Scaling
Brush management	Shrub encroachment rates vary with ecological site	Grass fragmentation with encroachment determines erosion rates	Mosaics of shrublands and grasslands can be conservation goals for wildlife
Invasive species management	Certain sites are more likely to be invaded first and serve as early warning	Formation of gaps in vegetation may facilitate initial invasion	When scale of infestation is large, it is difficult to control with herbicides
Prescribed burning	Fire prevalence depends on climate	Fuel connectivity determines scale of fire spread	Scale of fire-affected vegetation determines habitat values to wildlife
Prescribed grazing	Certain landforms or soils attract more use by livestock	Patch grazing breaks down with increased stocking rate	Increases in scale of pasture results in less uniform livestock distribution
Range planting	Species plantings more likely to be successful on certain soils	Plantings lead to reduced bare ground connectivity and runoff	Use of key microsites for establishment may lead to spread of seed to larger areas
Riparian management	Geomorphic valley type determines potential riparian function	Spatial pattern of channel (meander width ratio) determines riparian vegetation states	Upland vegetation management affects sediment deposition in riparian areas
Wildlife habitat management	Multiple plant communities may constitute desirable habitat	Fragmentation of forest leads to increased predation or parasitism rates	Dispersal limitation due to landscape fragmentation may override quality of local habitat

rangelands that included field data collection. This will allow detection of more subtle changes in ecological state or condition within rangeland vegetation in future evaluations, but only at very broad scales (http://www.ncgc.nrcs.usda.gov/products/nri/range/2006range.html).

There are two significant limitations to using existing NRI data to evaluate conservation practices. The first is that it is virtually impossible to obtain reliable information about the practices being applied at any particular random point. Although observable conservation practices are recorded, most of the more significant practices are not easily documented without expensive (and often unavailable) landowner interviews. Some of these data do exist, but they are housed in separate databases, are subject to privacy limitations, and generally lack sufficiently precise location information to relate them to field measurements.

The second limitation is that the sampling design was intended to provide estimates of change in major land use and land cover classes at regional to national scales. The design was not intended to detect differences among ecological sites or among variations in the arrangement of practices within a landscape. Such analyses require sampling that is structured with respect to the distribution of treatments and ecological sites in particular landscapes. Analysis must also consider particular models of interactions between locations in a landscape. Further, consideration of within-site heterogeneity may require sampling protocols that are tailored to specific processes (e.g., habitat use by birds). Thus, we assert that NRI is insufficient to provide an understanding of the effects of landscape context on conservation outcomes. This assertion suggests that additional conservation assessment strategies are needed at local to regional levels.

PLACING CONSERVATION PRACTICES IN A LANDSCAPE CONTEXT

Most chapters in this volume review literature to provide science-based evaluations of the effectiveness of individual practices and several of them point to the importance of landscape context. In the last 15 yr, the number of

rangeland studies that mentioned "spatial" or "landscape" increased from about 100 to over 700 per year, suggesting a growing awareness of the value of landscape perspectives within the rangeland science community. The number of studies referencing these terms that were associated with conservation practices, however, increased only slightly and is still a small number (< 40/yr). Consequently, this chapter does not on focus on specific practices, but instead offers a forward-looking synthesis of concepts and approaches that could facilitate a shift from the typically site-scale or local perspective toward a landscape perspective (Fig. 1). The local perspective uses information about the ecological state or plant community phase (plant community variants within states of an STM) to select practices to achieve target states or phases, sometimes tailored to an ecological site. The landscape perspective additionally considers how the surrounding landscape—including its states, ecological sites, and connections among them—influence the effects of practices. Attention is also paid to spatial heterogeneity or patchiness within the site as a regulator or objective of conservation effects. Finally, the landscape perspective considers how a localized conservation effect influences the landscape in which it is embedded. Below, we offer a review of several concepts underpinning the landscape perspective. Examples of how these concepts apply to practices discussed in other chapters are presented in Table 2.

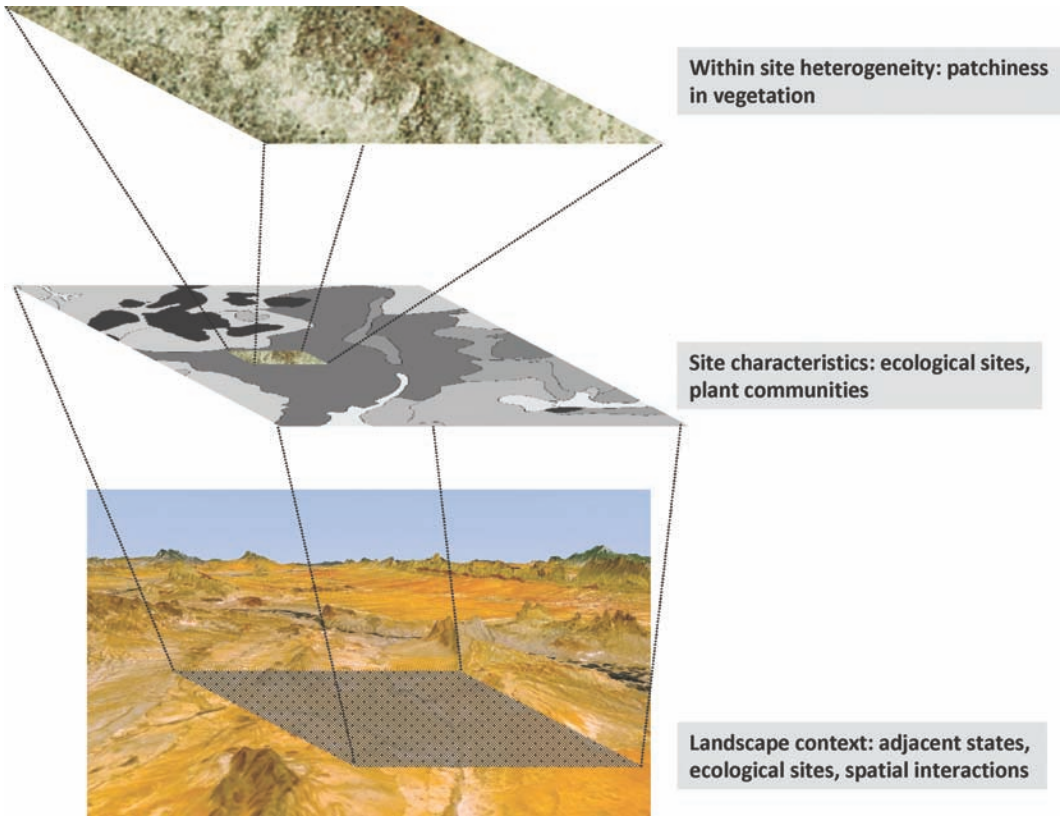
Spatial Heterogeneity

Spatial heterogeneity refers to the variability of properties over space including soils, vegetation, and process rates such as water runoff or erosion (Turner and Chapin 2005). Spatial heterogeneity within rangelands has been accounted for at some scales, but not others. For example, spatial heterogeneity due to relatively static variations in the landscape (reflected in the ecological site) and differences in past vegetation dynamics or "historical legacies" (reflected in ecological states or community phases) is known to influence the success of practices and is widely used in planning (Creque et al. 1999; Bestelmeyer et al. 2009). Different ecological sites or states/phases are predicted to differ in both the possible changes that can be observed and the success of certain management interventions. Different soils vary in their susceptibility to shrub invasion and the loss of perennial grasses, and in the potential for recovery of different grass species (Fuhlendorf and Smeins 1998; Hamerlynck et al. 2000; Wu and Archer 2005). For example, sites with well-developed argillic (clay-rich) horizons may limit invasion by both *Larrea* (McAuliffe 1994) and *Prosopis* (Archer 1995; Miller et al. 2001). On the other hand, the phytotoxicity of chemicals used to control shrubs may also be limited on clay-rich soils (Duncan and Scifres 1983). Once shrubs are removed, the recovery of grasses in response to removal may depend on clay content (Bestelmeyer et al. 2006a) and the degree of soil interspace erosion that is captured in classifications of alternative states (Herrick et al. 2006). The linkage of this type of information to ESDs and maps is potentially very useful. ESDs, however, often do not clearly present information on the likelihood of practice success or failure among ecological sites or states.

Spatial heterogeneity may also be of value in its own right, although considering heterogeneity in this way for planning is not common. It has long been recognized that habitat heterogeneity promotes biodiversity, but conventional measurements tend to emphasize within-habitat heterogeneity (such as the vertical complexity of vegetation at a point in space; Tews et al. 2004) rather than landscape heterogeneity (the composition and arrangement of different habitats across a landscape). Landscape heterogeneity in rangelands can be produced by the simultaneous and dynamic coexistence of different plant communities due to patchy disturbances and corresponding asynchrony in successional stages among patches. Such patch dynamics or a "shifting mosaic" (Bormann and Likens 1979) can yield desirable properties at broad scales. For example, shifting mosaics in grasslands caused by grazing and fire disturbances can create a mix of structurally simple and structurally complex plant communities that sustain wildlife populations that exploit resources in different plant communities over time (Archibald et al. 2005; Fuhlendorf et al. 2006). Spatiotemporal variation in vegetation can also be used as a tool to manage livestock distributions (Fuhlendorf and Engle 2004). Some valued wildlife species may even be associated with elements characterizing traditionally undesirable states, such as mesquite shrubs in semidesert

"…a forward-looking synthesis of concepts and approaches that could facilitate a shift from the local perspective toward a landscape perspective"

FIGURE 1. A schematic of the landscape perspective. The top panel illustrates patchiness in vegetation cover (dark) interspersed with sparsely vegetated areas (light). The middle panel is a map of ecological sites, each representing one or more vegetation states. The bottom panel illustrates the landscape within which the mosaic of ecological sites is embedded. These are three scales of pattern that promote a landscape perspective.



grasslands (Lloyd et al. 1998; Saiwana et al. 1998) or bare and structurally simplified areas in the Great Plains (Knopf 1996; Derner et al. 2009). Thus, the presence of even persistently altered states in parts of a landscape can have management value. In rangelands, however, management typically seeks to maximize the coverage of a single, desired plant community and deemphasizes landscape heterogeneity (Fuhlendorf and Engle 2001).

Spatial heterogeneity may be similarly important to the basic functioning of ecosystems. Shifting mosaics caused by localized fire and succession are seen as prerequisites for long-term sustainability of desirable forest structures (Allen et al. 2002; Bond et al. 2005). At a finer scale, alternating areas of bare ground that generate water runoff and vegetated patches that intercept runoff can sustain productive vegetation in arid environments where a more homogeneous distribution of vegetation could not be sustained (Noy-Meir 1973; Ludwig and Tongway 1995; Rietkerk and van de Koppel 2008). Consequently, conservation and restoration approaches to increase cover and production of desired species may focus on maintaining or creating heterogeneity and specific spatial patterns in vegetation (Noble et al. 1997; Miller and Urban 1999). In such cases, the existence of patches of bare ground or "early seral" vegetation may not reflect the initiation of ecosystem degradation, as is sometimes assumed, but rather are necessary components of some ecosystems that sustain productivity and biodiversity.

Spatial heterogeneity of the kind noted above is especially important for vegetation sampling. Point samples gathered without sufficient replication or stratification to different patch or community types can lead to misinterpretations about vegetation conditions when extrapolated

to the management unit or landscape scale. Sampling that ignores spatial heterogeneity and replication needs is a common source of monitoring failure (Herrick et al. 2005).

We conclude that explicit consideration of spatial heterogeneity in conservation planning and assessment is valuable because it contains information about practice effects. Spatial heterogeneity can also be a goal of practices. The information provided by spatial heterogeneity can only be obtained if our treatment of data moves from reporting averages or percentages to explaining the variability in the data as a function of ecological site, state, and patchiness variables. Such data can then be used to improve ESDs so that they better predict variations in success. Our concepts of ecological sites and states also need to describe spatial heterogeneity as an attribute of interest, rather than solely describing spatially averaged attributes and implicitly treating management for desired plant communities as an "all-or-nothing" proposition.

Spatial Pattern

Without direct consideration of heterogeneity, attributes such as soils, cover, and species composition become nonspatial. Spatial pattern within and among plant communities in a landscape, however, may have predictive value (Reitkerk et al. 2004; Barbier et al. 2006). Whereas spatial heterogeneity describes the condition of possessing dissimilar patch types in an area, spatial patterns communicate where those patch types are and how they are shaped and arranged. Spatial pattern can be measured by the density, size, shape, and location/adjacency of patches and plant communities, typically via a geographic information system (GIS), spatial statistics, and derived maps (Gustafson 1998; Turner 2005). Spatial patterns are descriptors of the potential for—and consequences of—spatial interactions involving the flow of matter (e.g., water, seeds, or animals), disturbance (fire or erosion), or information (cues for habitat selection). They thus communicate valuable information about specific ecological processes that affect the success of practices (Turner 2005).

Spatial patterns in vegetation give rise to variations in the availability of limiting resources or the intensity of disturbance. These variations subsequently alter the spatial patterns—a phenomenon sometimes referred to as "self-organization" (Watt 1947; Rietkerk et al. 2002). Self-organized spatial patterns are the consequences of feedbacks between the initial spatial pattern in vegetation and processes such as water redistribution. For example, a vegetation patch in a matrix of open ground will intercept the flow of water and increase infiltration and plant-available water resulting in increased plant growth. Increased plant growth may enable the patch to intercept more water and so on until resources from the adjacent bare area become limiting. As a consequence of feedback effects, rangeland landscapes often exhibit characteristic spatial patterns. As overall resource availability to the site changes (e.g., due to aridity), the spatial pattern may change in predictable ways (Rietkerk et al. 2004). Disturbances to patches that interfere with feedbacks also lead to predictable effects on spatial patterns and feedbacks (Ares et al. 2003; Kéfi et al. 2007). Large patches tend to become fragmented, leading to decreased resource capture, production, and soil degradation (Wu et al. 2000; Bestelmeyer et al. 2006b). The decrease in large patches also leads to characteristic changes in the distribution of patch sizes in an area. Thus, changes to spatial patterns have been promoted as early warning indicators of rangeland degradation (and presumably restoration success; Kéfi et al. 2007; Scanlon et al. 2007). The value of pattern-based indicators relative to standard measures of ground cover is being debated (Maestre et al. 2009).

The spatial pattern of ecological states, soils, and topography in a landscape governs the flow of resources and disturbances. The impact of practices on potential water yield is especially sensitive to the locations in a landscape where the practices are applied (Ludwig et al. 2005). For example, Wu et al. (2001) found that policy incentives for brush control on the Edwards Plateau need to clearly specify the optimal locations for treatment in order to influence water yield. Spatial specification is important because there are tradeoffs between strategies designed to increase potential forage productivity vs. water yield potential (Redeker et al. 1998). Similarly, the spatial design of infrastructure including the locations of

"Spatial pattern can be measured by the density, size, shape, and location/adjacency of patches and plant communities, typically via a geographic information system"

> “…the spatial pattern of vegetation cover can be even more important than the average amount of cover in determining runoff and erosion under some conditions.”

fences, watering points, and feeders are used to modify patterns of animal movement and forage utilization, taking into account livestock behavior and the template of topography and plant communities to which livestock respond (Laca 2009). In this way, the spatial locations of rangeland infrastructure can have a large, indirect impact on overall vegetation change. Similarly, the spread of fire in a landscape depends upon where the fire is initiated relative to the spatial arrangement of ridges and valley bottoms (Swanson et al. 1988) as well as the connectivity of fuel loads (Allen 2007).

The spatial pattern of vegetation patches resulting from practices, in turn, affects key processes and services. For example, Ludwig et al. (2007a) showed that the spatial configuration of vegetated and bare patches had a significant influence on erosion and sediment loss in northeastern Australia. A catchment with a coarse-grained patch structure (few large patches) and 54% grass cover had 43 times greater sediment loss than a catchment with fine-grained patch structure (many small patches) and 43% grass cover. This result suggests that the spatial pattern of vegetation cover can be even more important than the average amount of cover in determining runoff and erosion under some conditions. With similar amounts of total cover, the pattern of vegetation patches comprising this cover can be arranged such that they either slow runoff and retain sediment or allow it to leave the site. Larger bare patches, bare patches that are elongated parallel to the direction of flow, and bare patches that occur lower on a hill slope are less able to slow the movement of water and sediment and prevent its transfer into channels.

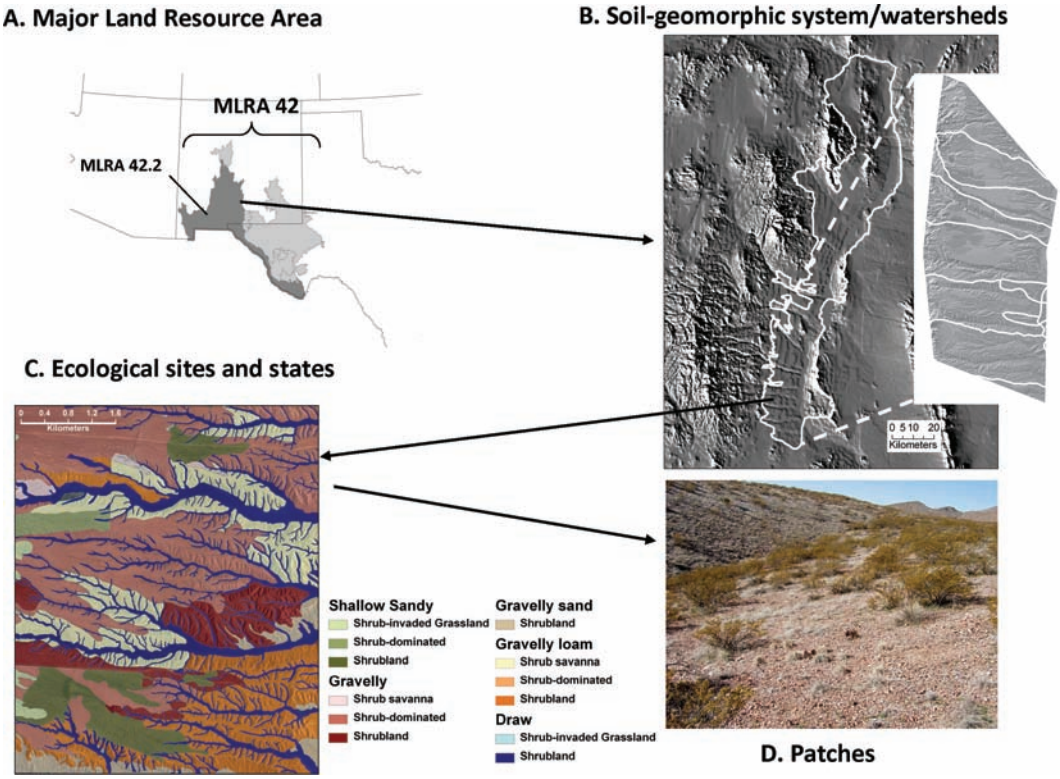
Water and sediment loss rates at sites with intermediate values of cover close to “percolation thresholds” can be very sensitive to changes in plant cover (Davenport et al. 1998). Percolation thresholds describe how a small change in cover over a defined area can result in a cover type becoming connected with respect to a process, such as the spread of fire (Turner et al. 1989) or the dispersal of species through certain cover types (King and With 2002). The shape and size of patches comprising cover affects the critical threshold value, but in general the shift from fragmented to connected occurs at intermediate cover values (Miller and Urban 2000). Areas with very low cover are necessarily fragmented and areas with very high cover are necessarily connected via a single large patch. Spatial pattern matters most when cover is intermediate because of the wide variety of possible arrangements of patches (Gergel 2005). Thus, attention to changes in connectivity can help us understand nonlinear relationships between cover and ecological processes in some conditions (Peters et al. 2004).

These examples suggest that various measurements of connectivity (and its converse, fragmentation) could be used to estimate critical ecological processes that mediate the effects of practices (Debinski and Holt 2000; Tischendorf and Fahrig 2000; Goodwin 2003). Some specific measures include the frequency of patches of different size or weighted mean patch size (Li and Archer 1997), the aggregation index (He et al. 2000), and the landscape leakiness index (Ludwig et al. 2002, 2007b) as well as several others (McGarigal and Marks 1995; Calabrese and Fagan 2004). As one example, we would expect that decreasing connectivity of vegetation patches and increased connectivity of bare patches would be reflected in a “landscape leakiness index.” Increases in this index are correlated with reduced water infiltration and nutrient retention. In selecting a metric, it is important to link it to a conceptual model of the pattern–process relationship, which, in turn, should indicate the appropriate spatial scale at which the metric is measured.

Biophysical Scaling Effects

The spatial extent of observation determines how we perceive natural resource problems as well as the practices we use to solve them. For example, woody plant encroachment in landscapes of South Texas was shown to be strongly scale-dependent. In areas encompassing multiple soils, woody plant cover was associated with high-clay soils and wetter portions of the landscape (Wu and Archer 2005). At finer scales within upland soils, woody plant cover was negatively related to soil clay content and was unrelated to surface hydrology (Archer 1995; Wu and Archer 2005). Thus, the correlates of woody plant dominance depend upon the spatial scale of measurement. Correlations with particular

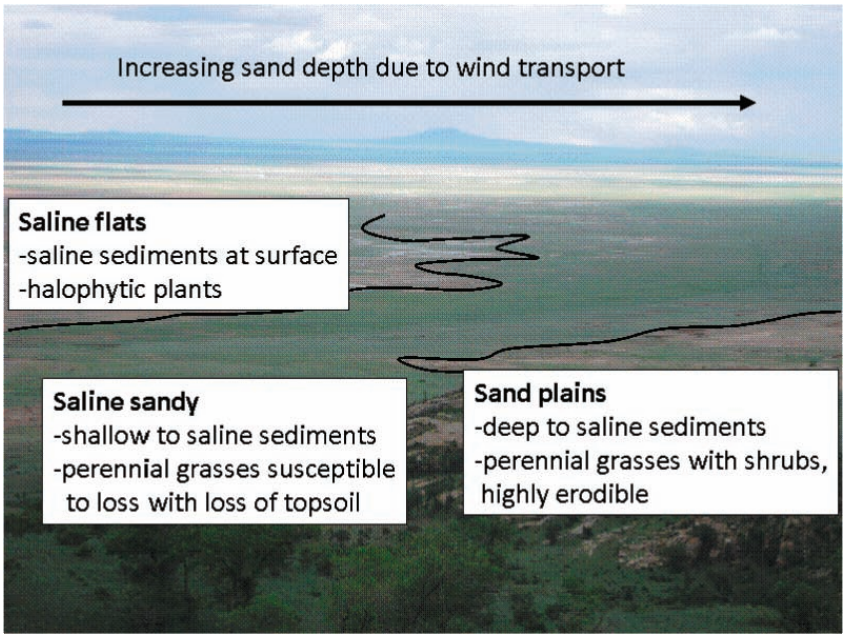
FIGURE 2. Example of a spatial hierarchy in a Chihuahuan Desert rangeland, following Table 2 and Bestelmeyer et al. (2009). **A**. Major Land Resource Area (MLRA) 42 and the Land Resource Unit MLRA 42.2, shaded. **B**. A soil–geomorphic system along the Rio Grande River valley comprising relict piedmont, ballena, and inset fan landforms associated with the Pleistocene–Holocene entrenchment of the river. The inset shows detail on soil map units and watershed boundaries (US Geological Survey hydrological unit code 11), illustrating the repeating patterns along the north–south axis. **C**. Each ecological site is represented by a different primary color. Various states occur within these delineations, including shrub savanna, shrub-dominated, and shrubland. States within sites are different shades of the primary color. **D**. Patches of *Bouteloua eriopoda* grass occurring in the interspaces of *Larrea tridentata* shrubs, a spatial pattern in the shrub-dominated state that indicates restoration of grass cover is possible.



variables (e.g., clay content) can change sign across scales due to interactions with factors that vary at broad scales (e.g., topographic position overrides the negative effect of clay on woody plant cover). Management models used to predict patterns of woody plant encroachment thus need to recognize the scale dependence of variables governing encroachment.

Different animal species or groups respond to rangeland attributes that are measured at different spatial scales. For example, although habitat quality for many bird species focuses on local vegetation structure, practices designed to promote highly mobile wetland bird species should focus on the distribution of a spatially dispersed mosaic of sites that are used at different points in the annual cycle (Haig et al. 1998). Considering the response of insect communities to grazing and mowing for hay in tallgrass prairie, Stoner and Joern (2004) showed that the species diversity of generalist and herbivore insect guilds in prairie fragments was largely controlled by local (within-fragment) plant community composition. This suggests that practices should focus on plant community attributes at the local scale to maintain populations of these insects. The predator insect guild, however, responded more to broader-scale attributes such as the shape of the fragments, thereby producing indirect effects on the other generalist/herbivore guilds. Attention to both fragment quality and fragment shape would be important conservation objectives in this case.

FIGURE 3. An example of ecological site concepts developed for an area near Dalinour Lake in the Inner Mongolia autonomous region near Xilinhot, China. The ecological site boundaries are defined by varying depths of sandy sediments over saline lake sediments that are associated with shifts in vegetation and vegetation change/soil processes. Because the three ecological sites are coevolving in response to a common process (wind-driven deposition of sand over saline lake sediments) they can be considered as part of the same soil–geomorphic system.



The spatial scale of observation also determines our ability to perceive processes that link the success of local practices to conditions in adjacent parts of a landscape. This perception is important when practices carried out at a fine scale influence adjacent areas due to spatial interactions. Conversely, the consequences of a practice within a local area may depend on influences from adjacent areas; both of these effects are termed "cross-scale interactions" (Peters et al. 2004). Clearly, some of the primary benefits of erosion control occur off-site in the form of decreased erosion in adjacent sites, improved water quality, and decreased siltation of streams and reservoirs (Pimentel et al. 1995; Pringle et al. 2006). Cross-scale interactions can also thwart local management efforts. Historical overstocking and drought initially converted extensive areas from perennial grassland to eroding shrubland until the 1950s in the Jornada Basin of southern New Mexico. Once the eroding shrublands became sufficiently extensive, many remaining grassland areas were converted to coppice-dune shrublands even when domestic grazers (and native grazers) were excluded via fencing (Peters et al. 2006). Grassland-to-shrubland transitions after the 1950s had become decoupled from the local processes that had previously caused them. Instead, they were controlled by broad-scale erosion and sediment movement that led to local soil instability with abrasion, burial, and mortality of grasses, occurring even in ungrazed areas (Okin et al. 2009). In such cases, the local management of vegetation or soils may not be adequate to predict the trajectory of vegetation change, as is often assumed in the use of assessment and monitoring indicators. A characterization of the functioning of the broader landscape would be required.

These examples indicate that in addition to variation in the properties of a specific land area (e.g., its ecological site or state) planners should carefully consider the landscape context within which an area is embedded. Furthermore, carefully chosen intervention points can induce nonlinear responses or emergent effects that are not predicted based on a simple linear scaling of the areas that are treated. One example might be to increase grass cover in critical portions of a watershed to reduce watershed-scale sediment loss. In both cases, there may be critical spatial scales at which the effects of key spatial interactions can be predicted, determined by factors such as geomorphology, hydrology, or species behavior (Turner et al. 2001). Our ability to predict these scales remains limited, but planners can integrate expert judgment with GIS to make informed (and testable) predictions.

Societal Heterogeneity and Scaling

Consideration of biophysical scaling will often lead planners to look to scales larger than a management unit. In these cases, the identity, heterogeneity, and spatial arrangement of management units in different ownership or tenure must be considered. Activities on one management unit may have off-site effects on adjacent units that are unrecognized, or diffuse effects from multiple units may influence attributes of communal interest, such as water table depth (Swallow et al. 2001; Standish et al. 2009). Collaborative approaches are thus necessary precursors to broad-scale practices such as fire management or species conservation (Sayre 2005). In order

for multiproperty practices to be successful, an atmosphere of trust and cooperation is required, calling for careful attention to an inclusive process in strategic and tactical planning (Duff et al. 2009). Much like the processes that lead to patchiness and sustainability in vegetation, "self-organized" groups of interested property owners working with agency representatives and scientists (e.g., prescribed fire associations) can lead to successful broad-scale conservation efforts (Biggs and Rogers 2003). Ultimately, however, successful practices must foremost be perceived to benefit individual landowners (Swallow et al. 2001). As with the use of ecological sites and states, it would be useful to document the societal contexts within which certain practices succeed or fail as a means of developing more effective approaches to conservation planning (Paulson 1998).

A SPATIAL HIERARCHY FOR CONSERVATION PLANNING AND EVALUATION

The preceding review makes a compelling case for the value of landscape perspectives in conservation planning and assessment, but how can we most effectively incorporate these perspectives? Some approaches, including the use of spatial simulation models, are too technically complex to be widely implemented at the present time. We suggest that informed judgment combined with GIS and selected use of some existing models (e.g., hydrologic models) provide a practical means to develop landscape perspectives. The concepts described here rely on spatial data. Such data can be used to detect patterns at different scales and then design and evaluate practices based on the patterns. Mapping activities—usually of management units, vegetation, and ecological

Water running in a rill after a rainfall event within a desertified grassland, southcentral New Mexico. (Photo: David Toledo)

TABLE 3. General levels of the land hierarchy discussed in this chapter, distinguishing characteristics for each level, approximate map scales, and analogous levels in the National Hierarchy of Ecological Units and Terrestrial Ecological Unit Inventory of US Forest Service. Entries in parentheses are not formal levels but are discussed in literature. This hierarchy mixes a spatial hierarchy (Major Land Resource Area to watershed) with elements of a classification hierarchy (ecological sites to patch) that can be delineated as nested spatial units.

General level used in this chapter[1]	Distinguishing characteristic	Map scale	USFS[2]
Major Land Resource Area	An area of similar gross physiography and continental weather pattern	1:3 500 000	Section
Land Resource Unit	A class or area based on regional climate variation or geology within Major Land Resource Areas; may or may not be spatially explicit	1:1 000 000	Section/subsection
Soil–geomorphic system	An area of similar geology and linked geomorphic/biotic processes that control landscape evolution	1:250 000	Land-type association
Watershed/Airshed	An area that is internally connected by a dominant spatial interaction (typically water flow, but could be eolian soil redistribution, fire, or animal movement)	~ 1:100 000, variable (hydrologic unit code 11)	(Watershed)
Ecological site	A class of land of similar potential vegetation, soil, geomorphic setting, topographic position, and microclimate	1:24 000 to ~ 1:150 000	Ecological type
Plant community/state	An area of similar plant species composition	~ 1:5 000 to 1:12 000	Plant association/ structural stage
Patch	A discrete unit of homogeneous vegetation and soil surface properties, ca. 1–100 m^2	1:1	(Patch)

[1]USDA NRCS (2003a) and Bestelmeyer et al. (2009).
[2]Winthers et al. (2005) and Cleland et al. (1997).

sites—are already part of planning and assessment process. Our recommendation is to plan and evaluate conservation practices with regard to multiple hierarchical levels of pattern in rangelands (Fig. 2) and forge more explicit connections with existing databases on soils and ecological sites. First, we describe a series of hierarchical levels in rangelands (from fine to broad scales) and the data that can be used to represent them (Table 3).

Patches

A patch is a relatively homogeneous area, often defined by local aggregations of plants or the absence of plants (e.g., a bare ground patch). Patch is a concept that can be used at any scale depending on the process or species of interest. The use of patch in this document is similar to the concept of the "pedon" used to describe a homogeneous unit of soil. Patch spatial patterns (the size, arrangement, and composition of patches) at fine scales (e.g., 0.1–1 ha) are used to define patterns within a plant community that affect specific processes such as erosion or habitat use. Information about patchiness serves three functions: 1) it changes our predictions when compared to the default assumption of uniformity (e.g., patchy vs. uniform grazing pressure), 2) it can be an objective of management (e.g., using fire to increase habitat heterogeneity), and 3) once recognized, patchiness can be altered to affect processes (e.g., to decrease landscape leakiness). For example, patchiness in grazing pressure can produce localized changes in vegetation that would not be predicted assuming a uniform grazing distribution, including changes considered to be useful or to be degradation (Adler and Lauenroth 2000; Augustine 2003). Patchiness produced by grazing, fire, or their

interaction has been promoted as a means to increase biodiversity (Fuhlendorf and Engle 2001). Practices can also take advantage of existing patchiness; erosion control structures can connect vegetation patches to form large obstructions to overland flow (see Ludwig et al. 1997; Reid et al. 1999). Slash additions combined with seeding may initiate the development of grass patches in eroded areas (Stoddard 2006) and the new grass patches may then expand over time.The alteration of patch spatial patterns can be measured using a number of tools. Ground-based transect approaches, including gap intercept, measure changes in the frequency distribution of fine-scale bare patches (Kuehl et al. 2001; Herrick et al. 2005). Aerial photography or high-resolution satellite imagery coupled to image classification are used to map vegetated patches (Bastin et al. 2002; Laliberte et al. 2004; Bestelmeyer et al. 2006b) and calculate a variety of patch metrics (Gergel and Turner 2001). More easily, Google Earth can be used to detect patch patterns across the globe and can now be linked to traditional GIS shapefiles (http://earth.google.com). Finally, the USDA Agricultural Research Service (ARS) Jornada Experimental Range has produced simple nominal and ordinal indicators that capture patch spatial patterns and associated soil redistribution processes (http://jornada.nmsu.edu/sites/default/files/FieldGuidePedodermPattern.pdf). The success of these approaches depends upon the spatial grain and the detectability of the patches involved. Measurements of patch spatial patterns (types, sizes, density, connectivity) could be explicitly defined as objectives (or preconditions) for state or Major Land Resource Area (MLRA)-level practice guidelines.

Plant Community

Plant communities, considered as spatial units, are assemblages of plant species and patches that exist at a particular place and time (Vellend 2010). Different communities in a landscape can be distinguished based on spatiotemporal shifts in the composition and abundance of species. Plant communities can be classified to states based on their responses to natural and management drivers and ecological sites may exhibit one or more plant communities (phases) or states (Bestelmeyer et al. 2009; Fig. 2). Practices are expected to produce or limit shifts among communities occurring in different states (accelerating practices) or produce shifts among communities within a state (vegetation management or facilitating practices; Stringham et al. 2003). Thus the identity of a community carries with it explicit predictions about its likely response to a practice.

In addition to the obvious role of plant communities as planning tools and assessment strata, data on the success of practices could be linked to communities and states in the Ecological Site Information System. These data can be used to refine STMs. Classifications of plant communities can also be linked to the responses of key animal species (Holmes and Miller 2010). Plant communities can be mapped using vegetation maps based on standardized vegetation classifications available through some gap analysis programs and other detailed mapping efforts (http://www.natureserve.org/prodServices/ecomapping.jsp). It is important to recognize, however, that coarse vegetation maps may combine several plant communities (and states) that are distinguished in STMs. Alternatively, plant communities can be mapped directly against a background ecological site layer using aerial photography or satellite imagery and derived spectral indices, often resulting in map units featuring associations or complexes of communities.

Watersheds in Kalalau valley, Kauai, Hawaii. (Photo: Brandon Bestelmeyer)

Currently, maps of plant communities are seldom available so they can be produced as needed for setting landowner objectives and implementing practices.

Ecological Site

Ecological sites are classes of land that differ in potential natural vegetation (Landres et al. 1999), historical range of variation, and response to disturbance as a function of differences in soils, landforms, and climate (USDA NRCS 2003a). The Terrestrial Ecosystem Survey of the US Forest Service provides land units similar to ecological sites (Winthers et al. 2005). Ecological sites are nested within climate-based classes called Land Resource Units (LRUs) or MLRAs (similar to ecoregions). In conjunction with its STM, the ecological site communicates the breadth of possible plant communities known to exist on a site. Even when STMs are similar, soil variations represented in ecological sites may influence the effects of management. Examples include the success of herbicide use with varying soil clay content, or variation in the success of grass seeding with climate variation among LRUs. Thus, planning and evaluation should be linked directly to the ecological site, and better still, to local information on soil/landform variations within ecological sites (Bestelmeyer et al. 2009). In this way, the classification of ecological sites can be updated to better reflect differences in ecological resilience or other responses, or the effects of important variations within ecological sites can be described.

Ecological sites are correlated to soil map unit components and are represented spatially via soil map units of the National Cooperative Soil Survey (e.g., http://soildatamart.nrcs.usda.gov). Given the scale of soils mapping in many rangelands, soil map units usually have a one-to-many relationship with soil components. As a result, soil map units describe soil complexes, associations or consociations that often translate to multiple ecological sites per soil map unit. For the purposes of initial stratification, many soil map units can usefully be classified according to the spatially dominant ecological site within them while recognizing that they are not necessarily homogeneous. Visual cues obtained in the field (e.g., surface soil color, gravel, slope) can be used to more precisely classify areas.

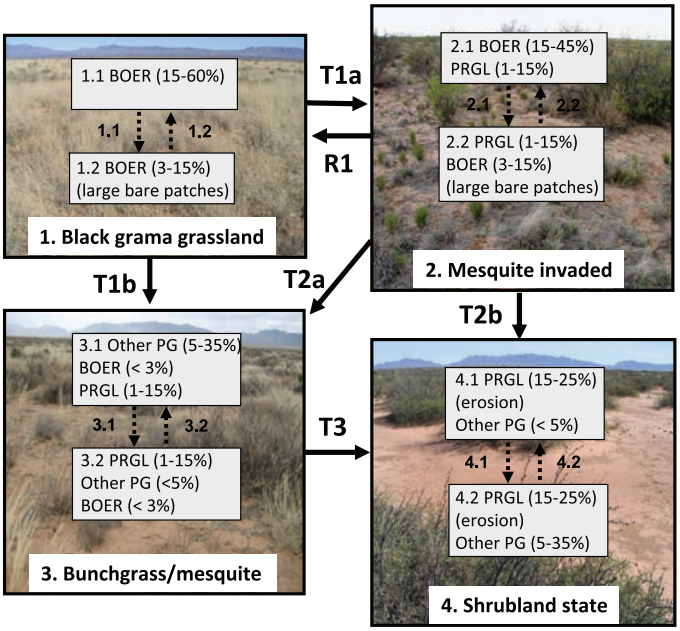
Watersheds/Airsheds/Firesheds

Ecological site and state units in many landscapes are connected to one another via hydrology and eolian transport or potential fire spread. Thus, management in one unit will impact others in connected landscapes. Watershed manipulations are often designed to take advantage of these connections; this has been reviewed elsewhere (Williams et al. 1997). To understand hydrological consequences, the appropriate order (or scale) of a watershed (i.e., Hydrological Units of the US Geological Survey) should be specified, alongside an expectation of the response to a hydrological manipulation at different scales and in specific parts of the watershed. Such predictions are more common within riparian zones compared with upland areas affecting riparian zones (Goodwin et al. 1997). Informal conceptual models or distributed hydrologic models can be used to develop such predictions. The concept of airsheds may be especially important in arid zones where wind erosion and sediment deposition processes are important (Okin et al. 2006). Airsheds could also assist prediction of the movement of smoke from prescribed fires. Similarly, "firesheds" have been conceived as the possible or expected area influenced by a single fire ignition as constrained by natural barriers, fuel, terrain, and weather within a given period of time (Stratton 2006). Formal units for the latter types of "-sheds" may not exist, but can be estimated from models or in a GIS. Accounting for physical connections is a key element in developing estimates of off-site effects of practices.

Soil–Geomorphic Systems(SGSs)

The SGS is a new concept and refers to a discrete land area with a characteristic spatial arrangement of ecological sites (and often plant communities) that are linked by fluxes of materials, organisms and disturbances, soil-forming processes, and ecological processes (Bestelmeyer et al. 2009; see Figs. 2 and 3). They are similar in scale to the landtype association of the National Hierarchy of Ecological Units (Cleland et al. 1997). Land areas within an SGS feature similar landscape organization and may encompass multiple watersheds or airsheds (depending on their scale). The interaction of management and soil or landscape attributes should be similar across an SGS. In other words, the rules

FIGURE 4. An example of a state-and-transition model developed for the Sandy ecological site, Major Land Resource Area 42 and Land Resource Unit 42.2. BOER = *Bouteloua eriopoda* (Torr.), Other PG = other perennial grasses (including dropseeds, *Sporobolus* spp.), and PRGL = *Prosopis glandulosa* (Torr.). Cover values (%) reported are foliar canopy cover from a variety of datasets. Key portions of state narratives relating to spatial patterns are highlighted to illustrate their use (right column). Specific predictions for transitions (T) and restoration pathways (R) occur below the figure.



State 1. Appearance of large (> 1m) bare patches signals risk of limited black grama recovery

State 2. Mesquite may preferentially invade in bare patches and then populations expand

State 3. Dropseeds die out in drought, ample opportunities for mesquite establishment

State 4. State spreads due to cascading soil erosion.

T1a. Mesquite establishment facilitated by seed transport by cattle, bare patches > 50 cm, and relatively wet springs
R1. Shrub removal with black grama recovery to > 15% may inhibit reinvasion
T1b, T2a. If black grama is reduced below ca. 3% cover (grazing in drought), it will not recover dominance
T2b, T3. At perennial grass cover < 5%, wind and storm events can trigger deep, spreading soil erosion

governing spatial interactions and determining the success or failure of a management action in a land area are similar within an SGS and will differ in distinct SGSs. Additionally, the spatial scales at which responses will be manifested, and should be monitored, can also differ among SGSs. SGSs can be used to tailor management and monitoring programs to landscapes that are structured differently. The extent of SGSs can be hand-digitized in a GIS using a digital elevation model and geology maps alongside a basic knowledge of hydrology and geomorphology or created by aggregating State Soil Geographic (SSURGO) database soil map units (sometimes State Soil Geographic (STATSGO) database map units can be used).

MLRAs and LRUs

Considerations at these scales are similar to those at the ecological site scale. It is useful to understand the location of an ecological site within a MLRA or LRU, given the continuous variations in climate that exist across the extent of these broad areas. Modeled climate products with national coverage, such as the PRISM model (Daly et al. 2002) can be used to quantify within-MLRA/LRU variations. The types of practices used and their outcomes vary strongly among MLRAs/LRUs, so it would be useful to assemble guidelines at these levels. For example, rangeland seeding has been recommended in the 10–14-inch precipitation zone (LRU) of MLRA 35 (Colorado Plateau) but not in the 6–10-inch precipitation zone.

A MODEL-BASED, LANDSCAPE APPROACH TO CONSERVATION PRACTICES

Using the spatial data and concepts described above, we recommend that the following steps be considered to design and test the effects of practices and then link what we

have learned from these tests to an expanding database. Planning starts with collaborative development of a conceptual model of the intended effects and ends with an update to the model, paralleling statistical approaches that are advocated for adaptive environmental management (Ellison 1996). In this case, the "model" includes recognition of spatial heterogeneity, spatial pattern, and landscape context.

Define Boundaries of the Management Area and Critical Natural Resource Issues

This is perhaps the most important step to incorporate biophysical and societal scaling. Knowledge of the primary conservation problems, the biophysical and social mechanisms involved, and the scales of spatial interactions associated with those mechanisms are used to delineate the spatial extent of a management area. General information about MLRAs, LRUs, and SGSs can be used to identify the types of mechanisms and critical scales that characterize an area (based on patterns in soils, geomorphology, hydrology, and climate) and therefore, the land area that needs to be considered to solve a problem. The extent of the management area alongside patterns of land ownership is defined in a GIS. Patterns of land ownership within the focal area will determine what resource concerns can be addressed on individual properties and therefore, which conditions can be expected to improve.

Collaborating stakeholders and planners then identify and prioritize natural resource problems and the specific locations of interest. The causes of the problems are identified, and historical perspectives on ecosystem conditions and drivers can help to recognize the key issues. Participatory mapping exercises (Reed et al. 2008) and workshops structured around general conceptual models of land change for an area (Reynolds et al. 2007) are useful approaches. The goal of this step is to focus limited resources on the highest-priority problems.

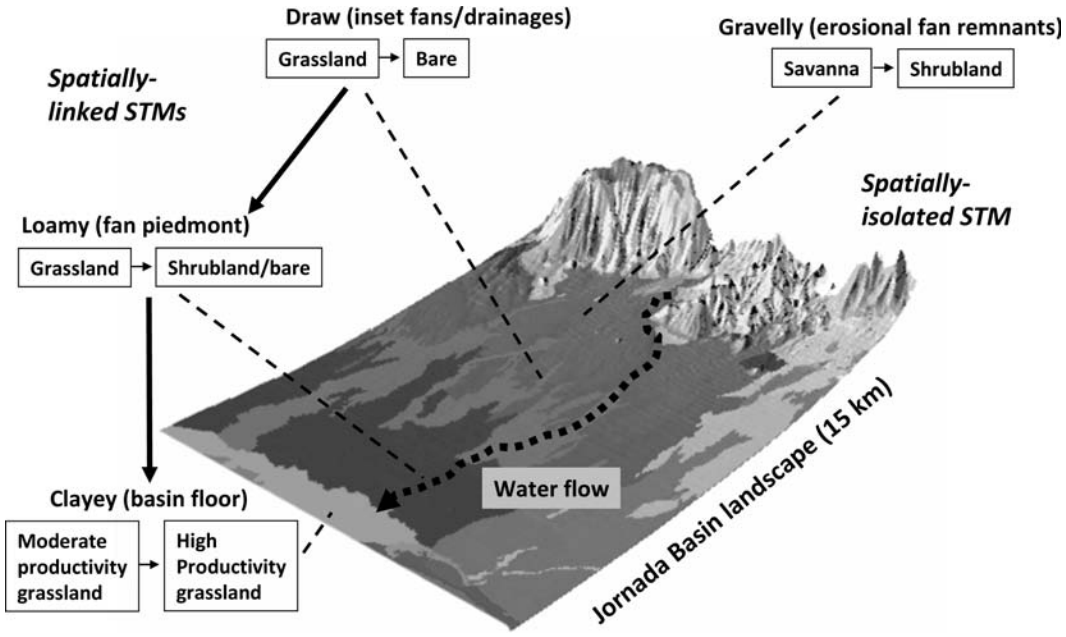
Develop Models of Conservation Effects

Soil or landform mapping is used to identify and locate the set of ecological sites present within the management area. This activity effectively stratifies the management area according to different conservation objectives and expected responses to practices. Each ecological site is then linked to a specific STM that describes the plant communities that are possible for each site and the drivers or interventions needed to achieve them. Ecological sites and STMs may already exist for the management area. Alternatively, ecological sites and STMs may need to be developed by project personnel in cooperation with NRCS and other agencies. Existing STMs must often be expanded to provide explicit predictions about practice effects.

A primary means to develop or expand STMs that serves the design of conservation practices is to examine historical applications and reconstruct their effects. This can be accomplished in many areas via comparisons of historical aerial photography and sometimes via ground-based data or photography. Local knowledge on how the practices were applied and information on the initial state and ecological site are essential. It would be useful to store information on past effects of conservation practices for each ecological site and state. Such a database does not yet exist at the national level, but the Land Treatment Digital Library provides a model for such a database (http://pubs.usgs.gov/fs/2009/3095) and similar databases could be developed locally. Additional sources of information include inventory data of the properties of plant communities associated with the same ecological sites and different management histories, recent monitoring data including responses to climate change and management interventions, and process-based studies that test for the mechanisms causing or constraining ecosystem responses, often associated with long-term research sites.

ESDs and STMs are used to subdivide the landscape according to conservation objectives and to specify the target states or plant community phases for each ecological site. A reasonable target depends partly on ecological potential, which depends on soil variations reflected in ecological site classification (e.g., the depth to saline sediments strongly affects the potential composition of plants; Fig. 3). The selection of targets and practices also depends upon either the risk of degradation or the nature of restoration thresholds that must be

FIGURE 5. A schematic of the relationships between state transitions (boxes and arrows) in state-and-transition models (STMs) associated with different ecological sites sharing a spatially interactive landscape. Transitions within the hydrologically isolated STM (Gravelly ecological site) have no effect on other transitions. A transition within the Draw ecological site has cascading effects downslope in the linked STMs. See text for additional explanation (from Bestelmeyer et al. 2011).



“Spatial data on states and ecological sites are the critical elements needed to connect predictions to specific sites and to assess spatial interactions across a landscape.”

overcome to achieve the target state. STMs can be used to formally define predictions about the responses of a discrete land area to conservation practices (Fig. 4). Most important, the STMs then become the repository for information learned from monitoring of practice outcomes. STMs linked to ecological site classifications function as evolving libraries and the interface between knowledge and action.

Although not generally available in existing STMs, local and landscape spatial patterns may be described as attributes defining at-risk community phases or alternative states (e.g., Ludwig and Tongway 1997; Fig. 4). For example, the presence of large open ground patches may signal an increased risk of invasion. Indicators of risk may also occur elsewhere in the landscape. For example, a head-cut gully several kilometers away might soon affect the vegetation and hydrology of an upslope area.

Identify Natural Resource Goals Across Ownership Boundaries

When conservation objectives suggest that cross-boundary coordination will be needed, stakeholders may differ in their preferences and perceptions of tradeoffs. Goals must sometimes be negotiated alongside building of trust between coordinating parties. Kitchen-table to community-level discussions and gradual consideration of the options are essential.

Develop Maps of Ecological Sites, States, and Landscape Models

Spatial data on states and ecological sites are the critical elements needed to connect predictions from STMs to specific sites and to assess spatial interactions across a landscape. Several tools currently exist to support this (e.g., Web Soil Survey, http://websoilsurvey.nrcs.usda.gov; Soil Web, http://casoilresource.lawr.ucdavis.edu/drupal/node/902). In relatively small areas, the solution is simple: conduct field assessments of vegetation and other state attributes (e.g., soil surface properties, patch patterns) alongside verification of the ecological sites. In large landscapes, we have used aerial photography and other layers (e.g., digital elevation models, soil maps)

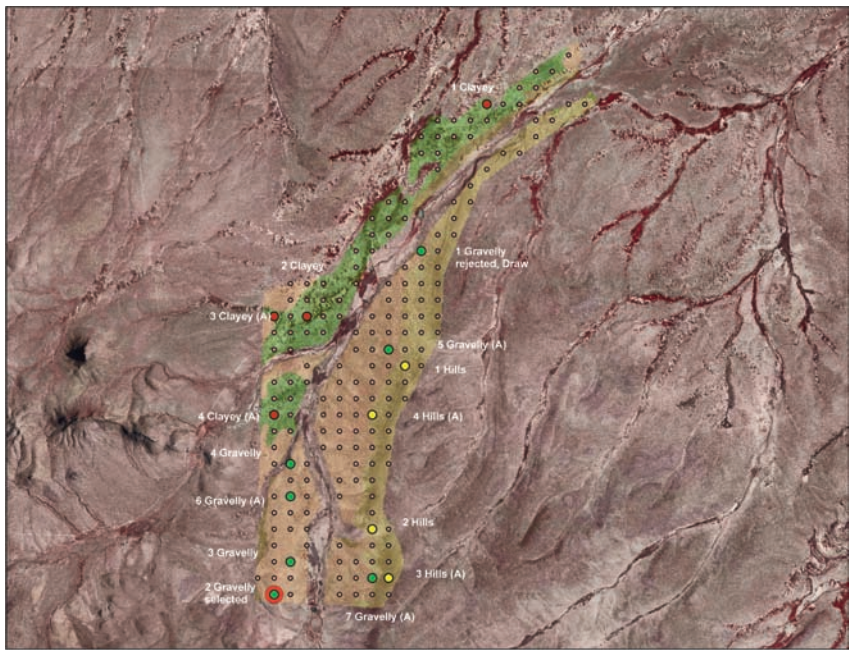


FIGURE 6. An example of a conservation effects assessment monitoring design within a brush management area treated with herbicide. The treated area was classified as having two ecological sites (green = Clayey, tan = Gravelly). Subsequent field assessments identified the Hills ecological site as an inclusion within one Gravelly polygon. ArcGIS was used to create a grid of systematically distributed points. A random number generator was used to select a subset of points for assessment of treatment efficacy and low-intensity monitoring (shrub mortality assessment and repeat photography points) in each ecological site (colored circles). One random subset point located in a nontarget ecological site (a draw that was misclassified in the spatial dataset) was rejected for use. An intensive monitoring point (outlined in red) in the spatially -dominant ecological site was randomly selected from the subset points at which standard monitoring procedures were employed.

in GIS to produce maps of ecological states and ecological sites (hereafter a "state map"). In using this approach in arid ecosystems, staff members at the Jornada Experimental Range have delineated map units that were believed to be internally homogeneous with respect to ecological site and state, but the identity of the state or ecological site was uncertain due to data limitations. Thus, they used the map to structure rapid field assessments and subsequently attributed the polygons. Assessments that were coupled to state mapping allowed a trained technician to evaluate the ecological site and state of 13 000–25 000 ha · day^{-1}.

The resulting maps can be combined with conceptual landscape models of likely spatial interactions or with GIS-based landscape models (e.g., Soil and Water Assessment Tool; Di Luzio et al. 2004) to predict how practices will influence broader scales via spatial interactions. As a simple example of the former, a planner can consider how STMs operating on different ecological sites may be linked within a watershed via a map of soils and a digital elevation model (Fig. 5; Bestelmeyer et al. 2011). Some models associated with hydrologically isolated landforms (e.g., Gravelly sites on erosional fan remnants, sensu Peterson [1981]) need not involve consideration of interactions with other models. On other sets of landforms, transitions among states are linked among ecological sites. A transition from a grassland to sparsely vegetated or bare state in a draw (inset fan) would result in a shift to shrubs and grassland species tolerant of drier conditions in a downslope ecological site (e.g., Devine et al. 1998). The sparser cover and increased runoff from this site, in turn, might lead to increased production at the edge of a Clayey basin floor and a shift from drier- to wetter-adapted grass species (e.g., Peters et al. 2006). Thus, in this example, a practice to repair the gully in the draw might result in both desirable and undesirable transitions in downslope ecological sites. This general set of interactions would operate throughout the SGS. In this way, the linkage of state and ecological site maps to landscape models can be used to delineate land units that require different practices and predict how they and adjacent units will respond.

Design Practices for Individual or Multiple Combined Land Units

For each land unit or for groups of land units that interact spatially (e.g., via hydrological connections), practices are specified to maximize the likelihood of successful maintenance of, or restoration to, the target state or phase. Where possible, an experimental component can be included using matched controls and pre-intervention measurements to allow a before–after–control-intervention statistical design (Block et al. 2001), preferably over suitable periods of time. Decisions are made for every land unit, including the decision not to intervene. The rationale for these decisions, based on the STM and stakeholder goals, can be stored with the state map database in ArcGIS software.

Monitor and Update ESDs

Monitoring stratified to different land units can test both the effectiveness of the practice application and the effects of the practice given its successful application. The monitoring program should be able to distinguish between application effectiveness and effect given successful application to provide fair evaluations of the causes of failure. Stratification by ecological state, ecological site, and surrounding states and sites allows context-dependent tests of practice effects. There should also be careful consideration of the hypothesized response attributes and timelines for change. Without careful consideration of these design elements, monitoring programs are often incapable of providing valid tests of practice effects.

In keeping with the collaborative nature of this approach, the interpretation of the monitoring data should be discussed among planners, science specialists, and stakeholders. Because the effects of intervention often unfold over long time periods and are influenced by short-term climate variability and other events, the results are sometimes not straightforward to interpret. The limitations of the data obtained at any given time should be recognized and interpretations can evolve with additional data.

The evidence obtained is used to modify or revise the appropriate STM, ecological site classification, and local landscape or general SGS models. As a result, the criteria for states and ecological sites may be changed and the likelihood of success of a practice within a state or ecological site can be quantified. The attributes of state maps can be updated and subsequent practices modified.

An Example: What Are the Benefits of the Model-Based, Landscape Approach?

The sequence of activities discussed above are new proposals, therefore we cannot provide direct evidence of their effectiveness. We can, however, provide an example of how they are currently being applied and the benefits we anticipate. The USDA ARS Jornada Experimental Range has worked with the Bureau of Land Management (BLM) in New Mexico to evaluate the effects of brush management practices that have been supported by both BLM and USDA EQIP funds as part of the Restore New Mexico program. Initial meetings with BLM staff were used to create explicit descriptions of the expected benefits of brush management. These meetings were also used to specify explicit hypotheses for vegetation responses in different ecological states and ecological sites. Digital maps were then used to design the brush control treatments. Soils, landform, and the pattern and cover of ecological states from aerial photography were interpreted according to STMs to identify suitable treatment areas within selected allotments. This selection procedure was based on interactions between BLM staff and grazing permittees. For example, shrubland states on thin, rocky soils are unlikely to yield much herbaceous response from brush control and were avoided in favor of slightly deeper soils. They used a simple spatial pattern indicator—the aggregation of perennial grasses under shrub canopies— to identify suitable states for treatment during field visits. They focused on states where remnant perennial grasses were distributed throughout shrub interspaces and in which vegetative recruitment could lead to rapid recovery (Bestelmeyer et al. 2009).

Once brush control treatments were applied, the same spatial data were used to design a monitoring program. Spatial data were used to stratify plots to target ecological sites and states and then randomly select plots within target ecological sites to achieve a spatially balanced sample (Fig. 6). They also established sampling plots in areas outside of treated areas on the same leased properties to evaluate changes to herbaceous cover that may occur when stock numbers are redistributed to other pastures due to grazing deferments in treated pastures. The size of the monitoring units (50-m transects) and monitoring methods were chosen considering the size and distribution of remnant grass cover patches. Line–point intercept, gap intercept, and belt transects (Herrick et al. 2005) are being used to monitor trends in herbaceous plant recovery and shrub mortality and recruitment, tailored to the expected responses of the brush control treatment. We have planned to obtain repeated readings over a 12-yr time horizon; desert grassland recovery is slow at best. Across the body of brush control

“planning data entered by field staff need to be integrated with spatial data and followed up by monitoring in treated areas as well as offsite areas”

treatments (over 50 sites), herbaceous response will be modeled as a function of soil properties, landform, landscape position, size of the treatment, weather, and posttreatment grazing management. Thus, this design integrates data on both local and landscape factors to design and test the practice. The need for local stratification, randomization, and measurements with respect to the specific treatments indicates the need for a carefully designed monitoring and spatial information system. The results of the Restore New Mexico monitoring will be used to update STMs for the area.

We anticipate that the model-based approach will produce results and benefits that have heretofore not been achieved. In spite of a long history of brush management activities, the lack of an ecological site and STM-guided experimentation and monitoring has circumvented a quantitative understanding of the conditions under which brush management succeeds and the characteristics of success. We anticipate that the Restore New Mexico monitoring will be able to more precisely target brush management activities to achieve desired results in the future. Model-based conservation planning might save millions of dollars that would be spent on ineffective treatments in southern New Mexico alone.

RECOMMENDATIONS

Incorporate Landscape Perspectives into Conservation Planning

How can agencies and conservation planners implement model-based approaches such as those discussed above? First, it is unreasonable to assume that all conservation planners will be able to access, manipulate, and store spatial data and process models underpinning the model-based approach. Expertise in GIS, remote sensing, and model implementation needs to be available to planners given a general knowledge of the uses of these tools. Such support within NRCS could occur initially via training and collaborations with other action and science agencies and with academic partners to make spatial data available in the planning process. The production of training materials and simple Web-based tools should be a priority. There should also be institutional support within NRCS to make expertise in GIS, remote sensing, and model implementation available to planners. Such support within NRCS could occur via spatial data specialist positions; similar expertise already exists in support of soil surveys.

Second, agencies alongside academic programs at universities should invest in longer-term training in landscape ecology and related tools and concepts, particularly GIS, hydrology, and soils/geomorphology. Most programs already emphasize elements of this training, but these elements are seldom integrated with ideas including ecological sites, STMs, monitoring, and approaches to specific rangeland practices. There is a clear need to develop integrative courses and texts that link the more disciplinarily specialized bodies of knowledge.

Third, administrative changes are needed in the development of conservation plans to include systematic consideration of off-site effects. Modifications to the Conservation Practice Physical Effects planning document to add an “off-site effect” category would be one approach. Historically, conservation planners incorporated landscape perspectives via rules of thumb such as “look across the fence to see what is coming at you,” but administrative requirements would ensure that spatial interactions are taken into account when needed. Finally, there must be an institutional structure within which STMs can be updated. This is a critical step if we are to learn from the study of conservation effects.

Develop Landscape Approaches to Conservation Effects Monitoring

Structured monitoring should be part of the budget for broad-scale conservation practices. Clear guidelines for design, institutional support for the implementation of the design, and a mechanism to incorporate what is learned from the monitoring within ESDs should be developed (as illustrated above). To accomplish this, planning data entered by field staff need to be integrated with spatial data and followed up by monitoring in treated areas as well as off-site areas. Careful monitoring design, including stratification and sufficient replication, must be supported if the monitoring is to be useful.

There should be a system in place to document the cumulative benefits of practices with

regard to large areas, long timeframes, and a broader range of services (Tanaka et al. 2005). Although there have been few attempts to project cumulative benefits of widely dispersed, small-scale restoration projects, Kondolf et al. (2008) developed a relatively simple approach to prioritizing projects to achieve maximum off-site benefits. This approach allows for an explicit statement of the assumptions about the delivery of ecosystem services offsite via the application of traditional site-specific management practices. Similar approaches merit closer consideration by NRCS.

Develop a Spatial Information Support System and Associated Tools

A spatial information system designed to support both the planning process and the design of monitoring programs requires a spatial database of maps and tools that can be used by planners, perhaps with assistance by spatial data specialists. The Web Soil Survey (http://websoilsurvey.nrcs.usda.gov) already provides a remarkable array of data and tools that can be used in the planning process and mirrors some of the steps described above. Soil Web, another recent online soil survey tool, links soil maps to Google Earth imagery (http://casoilresource.lawr.ucdavis.edu/drupal/node/902). It should be possible to expand Web Soil Survey and Soil Web by linking their data layers to others via Web-based distributed networks (e.g., climate from PRISM, http://www.prism.oregonstate.edu, or fire locations from GeoMAC, http://www.geomac.gov) and to project-level data housed in local servers.

The Landscape Toolbox (http://landscapetoolbox.org) provides an example of how various tools can be linked to specific

A restoration treatment using woody debris to create vegetated bands (patches) in eroded soils of Big Bend National Park, Texas. (Photo: Brandon Bestelmeyer)

Recently (left) and less recently (right) burned tallgrass prairie in east-central Kansas. (Photo: Brandon Bestelmeyer)

natural resource problems. The Landscape Toolbox employs an analytical framework to link specific problems to tools that provide information at different spatial scales. Projects such as the Landscape Toolbox highlight the value of linking disparate information sources and reveal the need for standards of information transfer (e.g., data formats and metadata) in the use of spatial data from multiple sources.

Maps that Facilitate Evaluation of Landscape Context. Such maps can be based on stable physical attributes derived from digital elevation models, including drainage networks, flow directions, and flow accumulation developed using GIS-based models. In addition to these more static attributes, climate data could be used to evaluate patterns of wind direction and velocity and fine-scale patterns of precipitation (e.g., via Doppler radar maps). Fire extents and characteristics can also be mapped. Readily available, preprocessed spectral data from the MODIS satellite can be used to document variations in production at landscape scales (e.g., maps of the Normalized Difference Vegetation Index with 250-m resolution). These data, in conjunction with visual interpretation of land surface characteristics in aerial imagery, can allow a trained specialist to recognize several important physical spatial processes.

Simple Spatial Models. Expanding upon the maps developed above, hydrological, fire or other models can be used to simulate landscape processes using digital elevation models, soil maps, vegetation maps, and other data as inputs (e.g., Stratton 2006). Such models

could be used to target, for example, particular portions of a watershed for treatment based on the sensitivity of water output to a change in plant cover expected in an area (Wu et al. 2001). More complex process-based models are being developed, but they are often designed for the purpose of exploring the effects of specific processes and it is unlikely that these models will be useful for guiding management anytime soon. On the other hand, simple models with user-friendly interfaces could enable the planners to develop alternative spatial designs for practices and compare the anticipated effectiveness of these options.

Spatial Pattern Indicators. Such indicators would be used to evaluate differences in patch or landscape pattern (e.g., patch size, patch density, connectivity, landscape leakiness) that are important mediators of ecological processes including water redistribution, erosion, or wildlife movement. Calculation of these indicators usually relies on classified satellite data or aerial imagery and the type of classification used depends on the process in question. Consideration of spatial pattern need not involve GIS-based calculations, however. Simple ground-based indicators for field inventory are available for upland (Tongway and Hindley 2004; Herrick et al. 2005; http://jornada.nmsu.edu/sites/default/files/FieldGuidePedodermPattern.pdf) and riparian areas (Prichard 1998). The specific indicators required will vary with the ecological process involved. As with vegetation composition, the measurements will change in response to practices and thus will be useful for monitoring. Assistance from spatial data specialists should be made available to support remote-sensing based monitoring. Protocols for the selection and use of specific indicators could be specified for MLRAs or groups of MLRAs that share similar ecological processes. Research support will be needed to identify useful spatial pattern indicators and to interpret their values. In this vein, reference values should be associated with descriptions of state and plant community phases.

Link Results to a National Ecological Site Database

Although the spatial information support system discussed above facilitates conservation planning, we must also consider where to house the monitoring data and the lessons learned from them. Similar to existing databases such as the National Soil Information System and the Ecological Site Information System, raw monitoring data at points and the interpretations derived from those data, respectively, will likely require separate, but linked databases. Current revisions to database structures planned with the reorganization of responsibilities for production of ESDs within NRCS provide an opportunity to consider how monitoring data on conservation effects could be linked to ESD interpretations. In any case, a database system and process to link CEAP monitoring data to ESDs must be a high priority within NRCS.

Support Research to Better Integrate Concepts, Tools, and Applications

Development of a systematic approach to conservation planning at the landscape level would benefit from research that addresses how to integrate information from landscape scales, spatial patterns, models, conservation planning field data, and NRI and other monitoring data. Specifically, such research would illustrate how field measurements should be gathered so that they can be scaled up or integrated with models and spatial data from broader spatial scales. Case studies centered on specific landscapes or MLRAs, supported by the USDA National Institute of Food and Agriculture and USDA Conservation Innovation Grants, could be used to explore how to bring together the variety of tools and approaches. Research is also needed to determine how best to scale up interpretations of conservation effects to state, regional, and national levels. Case studies illustrating how real-world conservation planning is linked to landscape research could provide an effective assessment of the benefits of the landscape perspective.

KNOWLEDGE GAPS

We identify the following knowledge and administrative gaps that need to be overcome in order move forward with our recommendations.

1. ESDs and STMs need refinement and elaboration so that they contain explicit predictions about how plant communities and dynamic soil properties are assumed

to change as a function of conservation practices. For predictions involving spatial interactions, levels that aggregate multiple ecological sites (such as SGSs) will need to be specified and carry the predictions. These predictions become the hypotheses for monitoring efforts and tests of them are used to update ESDs and refine our use of practices.

2. The lack of synthetic models at the level of MLRAs or LRUs is a clear limitation to developing consistent ESDs and STMs across the United States. Such models are needed to develop comparisons among different ecological sites and regions (e.g., the comparative likelihood of success or the magnitude of an effect in different land areas) and to represent spatial interactions at landscape scales (e.g., wildlife populations that cross multiple ecological sites).
3. Readily available maps of ecological sites, and especially ecological states, are generally not available to assist planning. Without maps that are connected to ESDs and STMs, planners will find it difficult to use these tools.
4. Models for spatial interactions in landscapes that specify how conservation practices in one state/ecological site mapping unit should affect the states of adjacent mapping units are poorly developed. This will require creative research and modeling coupled to field studies.
5. Spatial pattern indicators that aid in predicting the trajectories that states will take under different conservation practices are seldom available when they could be useful. Addressing this gap will require the integration of pattern analysis, using field- or image-based approaches, coupled to monitoring of conservation effects.
6. We lack administrative and database mechanisms to update ESDs and STMs. If the ESDs and STMs are not improved as a function of the monitoring tests, then learning cannot occur and the efficiency of conservation practices will not improve.

CONCLUSIONS

Our assessment indicates that landscape perspectives and applications are needed to promote long-term success and effectiveness of conservation practices on rangelands. A large body of literature supports the utility of a landscape perspective (e.g., Naveh and Lieberman 1984; Turner et al. 2001). We reviewed the implications of spatial heterogeneity, spatial pattern, and spatial scaling for the design of practices and interpretation of conservation effects. Spatial heterogeneity is used to understand why a practice succeeds or fails in areas of differing climate, soil, and spatial context. Spatial heterogeneity can also be a primary goal of practices, for example, by supporting the varying habitat elements used by animal species. Spatial patterns are used to indicate critical processes that are not reflected in other measures, such as connectivity for wildlife movement or runoff and erosion potential. Patterns too can be a conservation objective (e.g., wildlife corridors or areas of low landscape leakiness). Spatial scaling is used to understand the dimensions of the land area over which spatial interactions link practices in one place to effects in other places, and conversely, how characteristics of the landscape affect the local success of a practice.

Landscape perspectives encompassing spatial heterogeneity, pattern, and scale are increasingly being connected to practical tools that can be used by conservation planners. Such tools include indicators, classifications and maps of ecological sites and states, and hydrologic models. These tools can be used both to help design practices and to design the monitoring programs that evaluate their effects. A spatial hierarchy focuses attention on the data needed at each spatial scale governing ecological processes of interest. In order of decreasing scale, MRLAs, SGSs, watersheds, ecological sites, plant communities, and patches each relate to processes governing the management of rangelands. Furthermore, consideration of societal information such as land ownership is usually needed at broad scales. Each of these data sources can be consulted in a systematic way, which we described in six steps, to design and evaluate conservation practices in a landscape.

We recommend that conservation practitioners consider several scales of spatial pattern and related spatial processes, including cumulative effects, each

time a practice is applied. This synthesis indicates that a systematic approach to planning that incorporates landscape perspectives would, in many cases, lead to more effective interventions by 1) recognizing indicators foretelling the likelihood of success; 2) targeting interventions to ecological sites, states, and spatial contexts in which success is most likely; and 3) maximizing (and measuring) the cumulative, positive effects of practices over the long term at broad spatial scales.

Although some of the tools and approaches supporting landscape perspectives are already used by conservation planners, the development of others will require a scientific and institutional investment by the federal government and support by universities and funding agencies. Spatial data information systems should be developed that link maps, models, and pattern-based metrics to support planning and monitoring design. Databases are needed to house the resulting data. The interpretations of these data should be linked to ESDs. Foremost, we must invest in training and research to instill an understanding of the concepts and a capacity for reasoning about landscape processes (e.g., Gergel and Turner 2001). We suspect that such investments would pay for themselves, and then some, by improving conservation effectiveness in the millions of acres of rangelands that will be treated in years to come.

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CHAPTER

9

A Social and Economic Assessment of Rangeland Conservation Practices

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> The failure to include social and economic nonmarket values in decision-making and analysis will likely undervalue the net benefits of our nation's investments in conservation.

A Social and Economic Assessment of Rangeland Conservation Practices

9

John A. Tanaka, Mark Brunson, and L. Allen Torell

INTRODUCTION

Rangelands provide a wide variety of ecosystem goods and services, and the conservation practices implemented on them produce a variety of direct and indirect economic and social effects. Basic ecological relationships and varying degrees of natural resource management determine the magnitude and quality of goods and services produced. Society determines what the relative values of these goods and services are at any particular location and time (Fox et al. 2009).

In this chapter, we examine the literature related to the economic and social aspects of ecosystem services impacted by the conservation practices of the Natural Resources Conservation Service (NRCS) of prescribed grazing, prescribed burning, brush management, upland wildlife habitat, riparian management, and range planting. In addition, we examine the social and economic aspects of invasive species management that cross different conservation practices. At the time of this synthesis, invasive species management was not a specific conservation practice, but the NRCS recently created a new conservation practice titled Herbaceous Weed Management that is evaluated in a separate chapter of this document. Valuation of ecosystem goods and services potentially impacted by the specified conservation practices, particularly those services for which markets do not exist, is also examined. Understanding valuation methodology is important in evaluating conservation practice implementation and funding decisions. In some cases, the nonmarket valued ecosystem goods and services are those most valued by society.

The reason for estimating some measure of value for ecosystem goods and services is that landowners and managers need to evaluate trade-offs for decision making (e.g., Maguire and Justus 2008; Nelson et al. 2009). One way to make this evaluation workable is to put all the resources in the same units, and the assignment of monetary value is one way to accomplish this. However, the concept that an ecosystem good (e.g., an endangered species) or service has an intrinsic value that is "priceless" or "infinite" does not serve decision makers well when choices have to be made. The failure to include social and economic nonmarket values in decision-making processes will likely lead to undervaluing the net benefits and lead to inefficient allocations of our nation's investments in conservation.

The NRCS has recognized that ecosystem goods and services are directly and indirectly affected by the various conservation practices that they implement on rangelands. In the description of each conservation practice, the purposes describe the expected benefits or outcomes of practice implementation. Additionally, for each conservation practice, a physical effects worksheet is published that more specifically describes the benefits and outcomes. Both of these are on the electronic Field Office Technical Guide (http://www.nrcs.usda.gov/technical/efotg) sections of the NRCS website. This is shown in the descriptions of conservation practices and in the economic analysis of benefits and costs. In examining the conservation practice descriptions, there are a variety of different ecosystem goods and services listed as being positively or negatively impacted by the different practices. Table 1 shows a list of potential goods and services that can come from rangelands as currently recognized by the NRCS. As shown, there are many facets of each general good or service, each of which can have its own effect on the quality

Western Juniper expansion, Dufur Wildlife Management Area, Oregon. (Photo: John Tanaka)

TABLE 1. NRCS estimated impacts of different conservation practices on different ecosystem goods and services. 0 = not applicable, 1 = neutral, 2 = slight impact, 3 = moderate impact, and 4 = substantial impact. Parentheses indicate a negative impact. Adapted from NRCS Physical Effects Worksheets for each conservation practice (available at http://nrcs.usda.gov/technical/efotg as of March 2008).

	Brush management	Prescribed burning	Prescribed grazing	Range planting	Upland wildlife habitat	Riparian
Soil—erosion						
Sheet and rill	2–4	2–4	3–4	3–4	3	2–3
Wind	2–4	2–3	3–4	3–4	3	2–3
Ephemeral gully	2–4	2–4	3–4	3–4	3	2
Classic gully	2–4	2	2–3	2–3	2–3	0
Stream bank	2	2	2–4	2–3	2	3–4
Shoreline	2	2	2–4	2–3	2	3–4
Irrigation induced	0	0	2–3	0	0	0
Mass movement	(2–3)	0	1	1	2	2
Road, roadsides, and construction sites	0	0	0	0	0	0
Soil—condition						
Organic matter depletion	2–4	2	3–4	3–4	1	3–4
Rangeland site stability	2–4	2–3	3–4	3–4	0	2–4
Compaction	(2)	0	2–4	3–4	0	3–4
Subsidence	0	0	0	2	0	0
Contaminants						
• Salts and other chemicals	2–3	(2)	2–3	2	0	2–3
• Animal waste and other organics—N	0	2	2–3	2–3	0	2–4
• Animal waste and other organics—P	0	2	2–3	2–3	0	2–4
• Animal waste and other organics—K	0	2	2–3	2–3	0	2–4
• Commercial fertilizer—N	0	2	2–3	2–3	0	2–4
• Commercial fertilizer—P	0	2	2–3	2–3	0	2–4
• Commercial fertilizer—K	0	2	2–3	2–3	0	2–4
• Residual pesticides	(2)	2	2–3	2–3	0	2–3
Damage from sediment deposition	2–3	2	3	2–4	0	2–3
Water—quantity						
Rangeland hydrologic cycle	2–4	2–4	2–4	3–4	0	2–4
Excessive seepage	(2–3)	0	1	1	0	2–3
Excessive runoff, flooding, or ponding	2–3	2	2–3	2–3	(3)	(3)
Excessive subsurface water	(2–3)	0	2	1	2–3	2–3
Drifted snow	0	0	0	0	0	0
Inadequate outlets	2	2	2	2	0	0
Inefficient water use on irrigated land	0	0	(2)	0	0	0
Inefficient water use on nonirrigated land	2–3	0	2–3	2–4	0	0
Reduced capacity of conveyances by sediment deposition	2–4	2	2–3	2–4	2	2–3
Reduced storage of water bodies by sediment accumulation	2–4	2	2	2–3	2	4
Aquifer overdraft	2	0	1	1	0	0
Insufficient flows in water courses	2–4	0	1	2–3	2	2–3
Water—quality						
In groundwater						
• Harmful levels of pesticides	(2)	0	2–3	2–3	0	2–3
• Excessive nutrients and organics	0	2	2	2–3	0	4
• Excessive salinity	(2)	0	2	2	0	4
• Harmful levels of heavy metals	1	0	2	2	0	2
• Harmful levels of pathogens	1	0	2	2	0	2–3
• Harmful levels of petroleum	1	0	0	0	0	0
In surface water						
• Harmful levels of pesticides	(2)	0	2–3	2–3	0	2–3
• Excessive nutrients and organics	1	2–3	2	2	0	4
• Excessive suspended sediment and turbidity	2–4	2	2–4	2–4	2–3	3–4

TABLE 1. continued.

	Brush management	Prescribed burning	Prescribed grazing	Range planting	Upland wildlife habitat	Riparian
• Excessive salinity	2	0	2–3	2	0	2
• Harmful levels of heavy metals	2	2	2	2–3	0	2–3
• Harmful temperatures	1	1	1	1	1	2–3
• Harmful levels of pathogens	0	0	2	2	0	3
• Harmful levels of petroleum	0	0	0	0	0	2
Air—quality						
Particulate matter less than 10 µm in diameter (PM 10)	0	2–3	2–3	2–3	2–3	2
Particulate matter less than 2.5 µm in diameter (PM 2.5)	0	2–3	2–3	2–3	2–3	2
Excessive ozone	Neutral	1	1	1	1	1
Excessive greenhouse gas						
• CO_2 (carbon dioxide)	0	3–4	2–3	2–3	2	2–3
• N_20 (nitrous oxide)	0	0	0	0	0	0
• CH_4 (methane)	0	(3)	0	0	0	0
Ammonia (NH_3)	0	0	0	0	0	0
Chemical drift	(2–3)	0	0	0	0	0
Objectionable odors	0	(2)	1	0	0	0
Reduced visibility	0	1	2–3	2–3	2	0
Undesirable air movement	0	0	0	0	2–3	0
Adverse air temperature	(2–3)	(2–3)	0	0	2–4	1
Plants—suitability						
Plants not adapted or suited	3–4	3–4	3–4	4	3–4	3–4
Plants—condition						
Productivity, health, and vigor	2–4	4	4	4	3–4	4
Threatened or endangered plant species						
• Plant species listed or proposed for listing under the Endangered Species Act	1	1	1	1	1	1
• Declining species, species of concern	1	1	1	1	1	1
Noxious and invasive plants	3–4	3–4	3–4	3–4	3–4	3–4
Forage quality and palatability	3–4	4	3–4	4	3–4	3–4
Wildfire hazard	3–4	4	2–4	0	0	0
Animals—fish and wildlife						
Inadequate food	2–4	2–4	2–4	2–4	4	3–4
Inadequate cover/shelter	2–4	2–4	2–4	2–4	4	3–4
Inadequate water	0	0	0	0	0	2–4
Inadequate space	2–4	3–4	3–4	3–4	4	2–4
Habitat fragmentation	2–4	3–4	3–4	3–4	3–4	2–4
Imbalance among and within populations	2–4	2–4	2–4	2–3	4	2–4
Threatened and endangered fish and wildlife species						
• Fish and wildlife species listed or proposed for listing under the Endangered Species Act	1	1	1	1	3–4	1
• Declining species, species of concern	1	1	1	1	3–4	1
Animals—domestic						
Inadequate quantities and quality of feed and forage	304	4	4	4	2–3	3–4
Inadequate shelter	(2–3)	(2)	2–4	0	0	0
Inadequate stock water	0	0	0	0	0	0
Stress and mortality	2–4	2–3	3–4	2–4	0	3–4
Human—economics						
Land—change in land use	0	0	0	2–4	0	2–4
Land—land in production	0	3	0	4	0	2–4
Capital—change in equipment	3	2	2	2	2	2
Capital—total investment cost	2–4	2	0	3	2	4

TABLE 1. continued.

	Brush management	Prescribed burning	Prescribed grazing	Range planting	Upland wildlife habitat	Riparian
Capital—annual cost	1–3	1	1	0	1	2
Capital—credit and farm program eligibility	Situational	Situational	Situational	Situational	Situational	Situational
Labor—labor	1–3	2–3	2–3	2	2–3	2–3
Labor—change in management level	1–3	2	2	2	Negligible	2
Risk—yield	(3–4)	(2)	(2–3)	0	(2–3)	0
Risk—flexibility	(3–4)	3	2–3	2	2–3	0
Risk—timing	4	4	4	4	0	0
Risk—cash flow	2–4	(2)	(2–3)	2	2	2–3
Profitability—change in profitability	2–3	2–3	2–3	2	(2)	Situational
Human—cultural						
Cultural resources and/or historic properties present or suspected to be present	2–4	2–4	0	2–4	0	(2–4)
Human—energy						
Depletion of fossil fuel resources	No effect	(2–3)	0	0	2	(2–3)
Underutilization of nonfossil energy resources	(2)	2	0	0	0	0

and quantity of the good and service and its relative value to society.

Table 1 shows the expected change in the ecological or societal parameters from six main conservation practices evaluated in this document (invasive plant species management is not included). An examination of Table 1 indicates the ecosystem good or service followed by the expected level of impact from each of the conservation practices. While some goods and services listed are considered "bads," such as soil erosion (services can be either positive or negative, and those with negative outputs are called "bads" as opposed to "goods"; they are the outputs on which humans either place positive or negative values), the numbers indicate whether the conservation practice will minimize (positive numbers) or accelerate (numbers in parentheses) soil erosion. For example, in their critique of the ecological impacts of ranching, Freilich et al. (2003) identified some of the potential benefits that may arise from proper livestock management and also factors that need to be mitigated as those practices are implemented.

To put the values from Table 1 in context, Table 2 shows the number of hectares treated by each of the major conservation practices from 2004 to 2008 by state and rangeland region. Table 3 shows the conservation practice expenditures from all NRCS programs by state from 2005 to 2009. From an economic point of view, the scale of the practices being implemented over this period determines the potential size of the impact. Different states use some practices more than others (Tables 2 and 3) with expected differences in goods and services produced. The differences in expenditures by state and conservation practice (Table 3) may be due to a variety of factors, including local preferences, acceptability, and needs. Expenditures by state will also vary based on the amount of private rangeland and the willingness of those landowners to participate in NRCS programs. Expenditures in conservation practices from 2005 to 2009 (Table 3) in aggregate provide some indication of how practices have been implemented in different states. Table 4 shows the total annual expenditures for the conservation practices from 2005 to 2009. Since NRCS conservation programs are cost-share programs, these expenditures indicate only the government's share of the total investment in conservation. These figures also illustrate that federal expenditures have generally been increasing for these conservation practices over the 5-yr period. The question being asked is, "Do the net societal benefits, including both market and nonmarket values, from these practices offset the known costs?"

In this chapter, we discuss the various ecosystem goods and services impacted by these seven

TABLE 2. Area of the main conservation practices implemented in rangeland states from 2004 to 2008.

Location	Brush management (ha)	Prescribed grazing (ha)	Range planting (ha)	Prescribed burning (ha)	Riparian herbaceous cover (ha)	Upland wildlife habitat management (ha)
National	1 590 489	32 682 716	519 881	619 786	15 781	21 860 411
Rangeland region						
Central	1 046 181	16 780 643	257 038	336 161	8 630	12 185 583
West	410 656	13 815 577	260 263	34 660	3 722	6 980 085
West state						
Alaska	384	1 416 984	0	3	0	184 083
Arizona	34 413	2 607 496	5 228	3 148	59	332 011
California	28 491	588 229	6 924	6 197	511	152 638
Colorado	25 196	2 245 726	88 363	4 119	154	668 710
Hawaii	4 136	23 822	123	1 330	0	57 358
Idaho	8 996	239 791	3 359	1 790	193	180 574
Montana	37	1 046 954	94 561	9 406	31	735 365
Nevada	7 196	59 813	1 733	0	29	7 934
New Mexico	239 599	2 630 730	15 684	6 014	12	3 202 863
Oregon	12 061	290 328	5 006	987	2 619	222 809
Utah	27 561	423 301	37 417	140	3	217 394
Washington	5	65 912	433	103	74	124 209
Wyoming	22 581	2 176 491	1 432	1 421	38	894 138
Total	410 656	13 815 577	260 263	34 660	3 722	6 980 085
Central state						
Kansas	96 700	673 583	85 938	149 027	2	455 409
Nebraska	13 832	1 370 129	58 034	16 558	1 252	186 720
North Dakota	3 306	541 545	4 366	113	3 885	158 965
Oklahoma	170 900	1 247 742	32 143	84 726	645	303 599
South Dakota	1 709	343 747	2 894	302	2 140	158 251
Texas	759 734	12 603 897	73 663	85 436	706	10 922 639
Total	1 046 181	16 780 643	257 038	336 161	8 630	12 185 583

major conservation practices and determine if the peer reviewed literature provides measures of the quantity of change in the ecosystem good or service or merely considers those changes without quantifying them. We then evaluate what the literature indicates about the social effects associated with implementation of the conservation practices, the economic consequences of the conservation practices, and the economic valuation of the ecosystem goods and services. Our intent is not to specify what these values are today because they will change over time. Rather, we seek to define the ecosystem goods and services and their social and economic benefits as well as how those values may be used in decision making.

ECOSYSTEM SERVICES

Ecosystem goods and services are defined as those things or experiences produced by natural systems on which humans place value (Alcamo et al. 2005; Fisher et al. 2009; Fox et al. 2009; Kremen and Ostfeld 2005; Millennium Ecosystem Assessment [MEA] 2005). In this section, we examine the types of ecosystem services that may be increased or decreased from the implementation of conservation practices, identify major relationships among ecosystem goods and services, and describe the primary issues associated with measurement of those relationships.

TABLE 3. All NRCS program funds expended on conservation practices by state, 2005–2009.

Location	Brush management	Prescribed burning	Prescribed grazing	Range planting	Riparian herbaceous buffer	Wildlife upland habitat management	Total
West rangeland state							
Alaska	4 768	–	405 184	–	–	33 208	443 160
Arizona	2 896 930	17 256	4 573 259	326 474	–	65 166	7 879 085
California	3 311 387	15 270	1 045 000	928 945	13 104	92 985	5 406 692
Colorado	1 645 065	4 176	2 405 289	575 649	31	140 504	4 770 714
Hawaii	2 249 800	–	546 932	89 765	–	1 490	2 887 986
Idaho	270 053	4 372	929 023	349 916	2 013	7 199	1 562 577
Montana	83 108	–	3 266 740	722 150	588	4 196	4 076 782
Nevada	758 417	–	176 834	171 984	–	307	1 107 542
New Mexico	18 185 311	68 398	1 333 336	1 084 828	338	351 235	21 023 445
Oregon	1 735 163	97 104	3 195 510	524 326	–	150 912	5 703 015
Utah	1 087 580	2 152	773 954	860 948	525	20 790	2 745 949
Washington	219	–	149 788	145 726	2 264	191 632	489 629
Wyoming	717 663	21 132	2 950 802	48 960	–	67 070	3 805 627
West total	32 945 463	229 860	21 751 651	5 829 671	18 862	1 126 694	61 902 202
Central rangeland state							
Kansas	4 141 308	417 368	11 378 028	299 756	–	1 058	16 237 518
Nebraska	2 281 417	238 001	1 081 426	681 287	311	626 510	4 908 952
North Dakota	70 628	–	1 350 708	189 920	266	341 624	1 953 145
Oklahoma	10 301 158	766 992	1 067 643	691 442	–	64 273	12 891 508
South Dakota	6 664	–	609 149	351 212	–	71 877	1 038 902
Texas	77 507 317	436 684	9 585 612	5 720 478	4 024	44 434	93 298 549
Central total	94 308 492	1 859 045	25 072 567	7 934 094	4 601	1 149 775	130 328 573
Grand total	127 253 955	2 088 905	46 824 218	13 763 765	23 463	2 276 469	192 230 775

There are a variety of conceptual models used to organize and classify various ecosystem goods and services for purposes of informing management decisions (Ruhl 2008; Swinton 2008). The Millennium Ecosystem Assessment (MEA) is one such model (Carpenter et al. 2006) that sorted ecosystem services into provisioning (e.g., food, freshwater, fuel wood, and genetic resources), regulating (e.g., climate regulation, disease regulation, flood regulation, and erosion regulation), cultural (e.g., spiritual/inspirational, recreational, aesthetic, and educational), and supporting (e.g., soil formation, nutrient cycling, and primary production) categories. The conceptual model by Fox et al. (2009) provided the framework by which the ecological systems interact with the social and economic systems and defined the ecosystem goods and services as extractable goods and tangible and intangible services. While the MEA model has gained acceptance in defining these ecosystem goods and services, there seems to be a large amount of double counting that could occur if it were implemented on the ground. From a valuation viewpoint, double counting creates problems in summing up the total effects. The Fox et al. (2009) conceptual model did not seek to define the different ecosystem goods and services but instead made explicit that ecosystem goods and services provide the connection between human systems and their environment and hence their source of value.

Part of the issue with defining ecosystem services associated with implementation of

TABLE 4. Total annual expenditures through all NRCS programs for selected conservation practices in 2005–2009.

Conservation practice	Year				
	2005	2006	2007	2008	2009
West rangeland states					
Brush management	3 747 652	5 634 082	6 052 764	9 013 795	8 497 171
Prescribed burning	3 165	29 918	61 380	83 048	52 350
Prescribed grazing	3 205 190	3 731 971	5 842 649	5 116 415	3 855 425
Range planting	788 901	774 455	1 439 449	1 548 191	1 278 675
Riparian herbaceous buffer	7 882	3 989	2 635	3 430	927
Wildlife upland habitat management	30 415	95 562	210 273	328 152	462 292
West total	7 783 205	10 269 975	13 609 150	16 093 031	14 146 840
Central rangeland states					
Brush management	14 198 925	17 012 119	17 297 158	24 859 881	20 940 409
Prescribed burning	232 389	186 525	258 648	643 463	538 020
Prescribed grazing	1 802 474	3 400 594	8 407 342	5 071 747	6 390 410
Range planting	1 267 242	1 497 869	1 521 917	2 195 160	1 451 906
Riparian herbaceous buffer	64	694	—	3 688	155
Wildlife upland habitat management	123 393	173 081	244 566	365 460	243 274
Central total	17 624 488	22 270 883	27 729 631	33 139 399	29 564 174
Grand total	25 407 693	32 540 858	41 338 781	49 232 430	43 711 014

conservation practices is that there is little research on their production functions (Kremen and Ostfeld 2005). Production functions describe the relationship between the quantities and qualities of inputs used to produce various quantities and qualities of outputs. In addition, there is a need to understand the form of the relationship between different outputs described as the production possibility frontier by economists. For example, a given amount of input (e.g., land) can produce a variety of outputs (e.g., cattle vs. wildlife). Herrick (2000) concluded that we need to demonstrate causal relationships between soil quality and ecosystem functions such as biodiversity and biomass as well as the ecosystem's response to disturbance. Lal (2007) similarly presented arguments that soil science is the key if we are to pursue going to a carbon-based economy and meet the diversity of needs and wants from ecosystems. In any case, the characteristics of many ecosystem goods and services that will make them difficult to assess in decision making include their public good aspects, spatial and temporal dynamics, joint production, complexity of ecosystems, interdependence benefits, and the interactions among these characteristics (Fisher et al. 2009).

There are many pressures on the production of ecosystem services originating from agricultural and natural resource management, including conservation practices, and societal issues, such as urbanization and land fragmentation. Converting land use from the production of typical agricultural products to the production of biomass for energy can also drastically alter the quantity and type of ecosystem services produced on a given land tract. Cook et al. (1991) estimated that the potential exists for 20 million ha of US rangeland to be converted to energy-producing biomass with impacts on wildlife habitat, soil erosion, salinization, groundwater depletion, and subsidence. The study did not, however, quantify the expected changes in those ecosystem services. Higgins et al. (2002) looked at the potential impacts of agricultural practices and development as threats to future waterfowl habitat conservation over time. They concluded that changing economic and policy pressures on farmers and ranchers have the potential to modify management practices to bring marginal land currently in the Conservation Reserve Program (CRP) back into crop production.

In a study to evaluate the effects of western juniper (*Juniperus occidentalis* Hook) control

on an eastern Oregon ranch, environmental services were included in the ranch model to evaluate their response to juniper control (Aldrich et al. 2005). While this study did not estimate the production function for environmental services, it did show how erosion and wildlife populations may change in response to the implementation of different management alternatives. Wildlife species (quail, deer, and elk) responded differently because of unique habitat needs as the percent canopy cover of juniper changed from the alternative juniper control practices and the implementation timing. Erosion potential decreased as the trees were removed and cover of grasses and shrubs increased. Other models have been used in similar situations to evaluate tree control to increase forage production (Engle et al. 1996). Unfortunately, the manner in which various responses of ecosystem services might impact landowner decisions was not considered in either study.

A comparison of cattle production with quail and deer habitat in Oklahoma indicated that ranch returns varied based on the amount of wildlife present and the various brush management treatments that were used (e.g., prescribed burning, herbicide applications, and mechanical treatments; Bernardo et al. 1994). Lease hunting showed higher net returns per hectare compared to cattle production when wildlife populations were abundant but with no lease hunting income.

The development of a production possibilities frontier (PPF) requires knowing how each ecosystem good or service responds to common inputs such as vegetation (McCoy 2003). A PPF was developed to compare cattle and antelope in Wyoming (Bastian et al. 1991); however, the PPF did not show a great deal of substitutability because dietary overlap between the two species was minimal. It was concluded that it would take extreme value differences between cattle and antelope for the economically optimal combination of species to be only cattle or only antelope.

A PPF describing the effect of cattle grazing on carbon and nitrogen balance of mixed-grass rangelands was developed comparing light, heavy, and ungrazed pastures (Schuman et al. 1999). Aboveground biomass, carbon, and nitrogen showed a curvilinear (decreasing at an increasing rate) decline with increasing grazing intensity. When considering both above- and belowground biomass, a U-shaped curve was observed where total carbon decreased at an increasing rate and total nitrogen was about linear because of grazing intensity. In these kinds of studies, three treatment levels can begin the process of identifying the PPF curve in order to assess the trade-offs among unique combinations of the two products. More treatment levels will lead to better decision-making capabilities, allowing for more finely defined points to ascertain continuous trade-offs.

The quantity and quality of forage produced is one of the major ecosystem goods that is valued in the market, which makes it relatively easy to value compared to other ecosystem goods and services (Bartlett et al. 2002; Council for Agricultural Science and Technology 1996). However, forage production can have other values beyond that which is placed on it in the marketplace. The market value of forage tends to be heavily weighted toward domestic livestock production, but it can also have other values, such as wildlife feed and habitat, erosion control, and quality of life. These additional values are likely to be captured only through nonmarket valuation methods. It would be necessary to estimate a livestock production value, a wildlife feed and habitat value, an erosion control value, a quality-of-life value, and social benefits and costs if the total economic value for forage is to be estimated (Bartlett et al. 2002).

Implementing conservation practices can have the unintended consequences of reducing some ecosystem goods and services. For example, it is generally viewed that controlling salt cedar (*Tamarix* spp.) is a desirable practice when the objective is to improve riparian habitat for numerous wildlife species. However, as Dudley and DeLoach (2004) pointed out, when an endangered species such as the southwestern willow flycatcher (*Empidonax traillii extimus*) uses salt cedar for habitat, controlling salt cedar presents the problem of an incidental taking of an endangered species even though the native vegetation might eventually provide better habitat. In addition, salt cedar or its control can have impacts on water availability, other

Small acreage owners in Utah County, Utah, maintain livestock that can affect surrounding agricultural land. (Photo: Mark Brunson)

wildlife, aesthetics, and forage availability, resulting in multiple trade-off decisions. Beyond these trade-offs, one of the main reasons to control invasive plant species such as salt cedar is to prevent its eventual spread to other areas. The bottom line for such decisions are both spatial (e.g., water for downstream use vs. wildlife habitat at the point of control) and temporal (spread over time).

In a Texas study of CRP lands, a comparison of targeting only high potential programs based on costs, benefits, or the benefit-to-cost ratio found mixed results for environmental benefits (Babcock et al. 1996). They found that using the proper criterion (benefit to cost) can result in greater environmental benefits, including reduced wind and water erosion, increased surface water quality, or better wildlife habitat, compared to other selection criteria for enrolled lands. They noted that heterogeneity of environmental quality and productivity affects the magnitude of changes in the environmental effects. They were not able to estimate the impact on the production quantity for many ecosystem services because of the lack of ability to quantify physical trade-offs among the ecosystem services and the absence of a social value function to evaluate societal trade-offs.

It may be possible to increase ecosystem services through changes in management options. Integrating crop and livestock systems in Texas was shown to improve nutrient cycling, reduce soil erosion, improve water management, interrupt pest cycles, and spread

Salt Cedar (tamarisk) control, Bosque del Apache National Wildlife Refuge, New Mexico. (Photo: John Tanaka)

economic risk through diversification (Allen et al. 2008). Grazing was shown to increase soil organic carbon and nitrogen contents with light grazing compared to no grazing or heavy grazing (Ganjegunte et al. 2005). Brush management may be a way to increase water yield as well as bird habitat for species that require grasslands (Olenick et al. 2004b).

In Arizona, an estimate was made of the value to home owners from riparian habitat restoration, and it was concluded that the benefits exceeded the costs in this case (Bark-Hodgins and Colby 2006). Although the value is not solely attributable to riparian restoration, it does identify hedonic pricing models (essentially a regression model that uses characteristics to explain differences in price) that relate numerous attributes to the value of the land as one method of estimating the value to property owners of some ecosystem goods and services. A similar hedonic model was used to estimate amenity values for agricultural land in Wyoming. Land that could offer several economic services, including scenic views, elk habitat, sport fishing, and distance to a nearby town, was shown to command a higher selling price (30%) than similar land that did not provide these ecosystem services (Bastian et al. 2002).

The relationship between the type of ecosystem being improved and the management uses of the riparian zone affect how a conservation practice can be implemented to improve riparian vegetation. Quinn et al. (2001) developed a relationship between a riparian zone classification and the potential for riparian zone improvement in ecological health. Implementing practices such as off-stream water development to draw cattle out of riparian areas can have beneficial effects on both livestock and the riparian area (Stillings et al. 2003). Changing the status of riparian habitat was shown to have differential effects on amphibians, reptiles, birds, and mammals (Ekness and Randhir 2007). They found that the higher the degree of disturbance to the riparian area, the greater the negative impact on these four wildlife groups. They concluded that spatial targeting of conservation practices will have the greatest positive effect when targeting headwaters and lower-order watersheds. They also developed a conceptual model to evaluate the role of conservation practices that affect watershed characteristics important to the wildlife groups that they studied. Freemark (1995) developed a spatial–temporal hierarchy to illustrate the scales at which conservation practices and other stressors can affect wildlife in agricultural landscapes. The implication is that at these different scales, conservation practices may have differential impacts on wildlife habitat and populations.

In this section, we have sought to define and identify the kinds of ecosystem services that can be expected to arise from the implementation of conservation practices. The basic premise is that these ecosystem services arise, either intentionally or unintentionally, from the conservation practice and can have either a positive or a negative value. Understanding the relationships and relative values of the ecosystem services is crucial for making investment allocation decisions and to determine whether an investment in a conservation practice is going to be profitable.

ECONOMICS

Prescribed Grazing

The NRCS conservation practice standard for prescribed grazing (US Department of Agriculture [USDA]-NRCS 2007) defines "prescribed grazing" to be the controlled

harvest of vegetation with grazing animals, managed with the intent of achieving a specific objective. Sustainability of forage and livestock production are central concerns when designing a grazing strategy, but there is a body of literature dealing specifically with the economics of grazing. Choosing an optimal stocking rate is considered one of the most important grazing management decisions because the stocking rate decision affects vegetation, livestock production, wildlife and economic returns (Holechek et al. 2004). We first describe the economic model typically used to define economically optimal stocking rates and then review the literature dealing with the prescribed grazing conservation practice.

Optimal Stocking Rates. The stocking rate decision is a classic example of the well-known production economic model of profit maximization when defined from the input perspective (Debertin 1986; Workman 1986). The traditional myopic single-year economic model ignores potential interyear grazing impacts and equates the added economic value of an additional grazing animal to the added cost of that animal, a principle commonly known as equating value of marginal product to marginal factor costs (VMP = MFC). With diminishing rates of gain as more animals are added to the pasture, each animal added contributes less to profit than did the previous one, and at the economically optimal stocking rate, the last animal adds nothing to profit.

The conceptual economic model was described over 45 yr ago by Hildreth and Riewe (1963) and has been applied primarily to yearling stocker cattle because of the added complexities of cow–calf production (but see Hart et al. 1988b). Regardless of the animal class, the expected production rate for grazing animals is related to the number of animals grazing a given land area. Based on declining per head performance, a production function is defined that relates the gain per hectare that would be realized at alternative stocking rates. The principle of diminishing returns applies and animal gains will eventually decrease as stocking rate is increased. Total gain per hectare will also eventually fall, but this may occur at a stocking rate that is well beyond what would be detrimental to rangeland condition and future forage production. In this case, rangeland condition and sustainability over time become of key importance, and a dynamic economic model is needed. However, given no major year-to-year interactions, the single-year economically optimal stocking rate will lie somewhere between the relatively low stocking rate that would yield the biggest calf and the relatively high stocking rate that would give the most gain per hectare (Torell et al. 1991).

A search for literature in AGRICOLA with screening on rangelands and the search term "stocking rate" in the title or subject field, with "economics" and "rangelands" in the key words, identified 156 papers, of which no more than about 40 actually did an economic assessment of stocking rate alternatives. Numerous studies applied some variation of the single-period model of optimal stocking rates as described by Workman (1986) with notable examples including research conducted in Wyoming by Hart and various coauthors (Hart et al. 1988a, 1988b; Hart 1991; Manley et al. 1997). Hart's model applications improved the economic assessment of optimal stocking rates by modifying the input to be the number of animals grazing per unit of forage produced (grazing pressure [GP]) and not animals per hectare (stocking rate [SR]). The Hart studies were also unique in that long-term grazing studies were used to define key production relationships. Most economic studies about stocking rates and rangeland investment analysis have typically used biophysically simulated data (Huffaker and Cooper 1995; Aguilar et al. 2006; Teague et al. 2008).

Hart's revised definition of grazing input showed that a given cost–price situation results in an economically optimal GP for the current grazing period, but the optimal number of animals stocked per unit area (SR) is also determined for the given forage condition. The economically optimal stocking rate will depend on sale prices and production costs and will vary annually. Further, the production function captures key input–output relations crucial to the economic assessment. A favorable rainfall year means more forage, and the production function shifts upward. Rates of gain will be different for different grazing seasons, and the economic assessment is defined for a particular grazing season and grazing system. An altered grazing season or rotational scheme potentially

Grazing in Northeast New Mexico. (Photo: Victor Espinoza)

Low sagebrush grassland, Indian paintbrush, Big Horn Canyon National Recreation Area, Wyoming. (Photo: John Tanaka)

shifts the production function up or down. A limitation is that the traditional economic model does not include the economic value of other ecosystem services, and it considers production only during the current period.

Because of the complexity and lack of response data, few economic studies have moved beyond the simple single-period economic model of stocking rates. Yet, for the cow–calf producers most commonly using western rangelands, interyear interactions always occur, and the stocking rate decision becomes much more complex with the additional uncertainty about available forage now and in the future. The cow herd must be maintained across years, and forage availability is highly variable between years. Overgrazing during dry years becomes problematic and controversial when conflicts arise between livestock producers and land agency personnel about reducing stocking rates during drought periods.

Early dynamic economic studies of optimal stocking rates included Burt (1971), Karp and Pope (1984), Pope and McBryde (1984), Rodriquez and Taylor (1988), Garoian and Mjelde (1990), Torell et al. (1991), and Huffaker and Cooper (1995). There is a widely held belief that individual short-term optimization is at odds with long-term sustainability of an ecological–economic system, suggesting that a dynamic approach is needed. Several studies have not found this to be the case, however. Torell et al. (1991) found the *intertemporal* grazing impacts on forage production were not that important. If profit-maximizing livestock producers would maximize profit during the current period, then nearly identical stocking decisions would be made as those obtained from a dynamic decision model, and optimal stocking rates would be at sustainable levels. Falling animal performance was the critical driver for stocking rate decisions. The number of stocker animals in

the pasture would optimally fluctuate annually with forage conditions, beef prices, and production costs. Only occasionally would the cost–price situation be such that major interyear forage impacts would occur (Torell et al. 1991). Similarly, Quaas et al. (2004) concluded that for typical semiarid rangelands, under plausible and standard assumptions, short-term optimization leads to sustainable outcomes. Contrary to this finding, Teague et al. (2009), using a simulation model of semiarid savanna rangeland, found that sustainable stocking rates were 67–75% those that would maximize profit to livestock producers. They note that earning potential was four times higher for range in excellent condition and suggest a need to stock rangeland more lightly so as to prevent rangeland degradation and improve range conditions. The risk of potential herd liquidation and the need for feed purchase along with other negative impacts to the rangeland resource increase as stocking rates increase.

Matching stocking rates with dynamic forage conditions has emerged as the most consistent management variable influencing both plant and animal responses to grazing (Briske et al. 2008). It follows that it is also the most important factor influencing ranch profitability and economic responses to grazing. Typical drought management strategies include increased supplemental feeding, maintaining a conservative stocking rate so that destocking is rarely necessary, maintaining grazing flexibility by having yearlings as one of multiple enterprises on the ranch, and leaving a significant amount of herbaceous production at the end of the grazing season (Stafford Smith 1992; Hart and Carpenter 2005). Torell et al. (2010) found that interyear forage variability decreased net ranch returns by 46% relative to what could be obtained without variable forage conditions.

Economics of Grazing Systems. A deferred, rotational or other type of grazing system must result in one of two production responses for the practice to be economically beneficial to a livestock producer. First, animal performance must be improved with the alternative grazing system (shifting the production function up), or, second, forage production must improve over time (shifting the production function up gradually over time). It is also possible that these grazing systems may support more effective management decisions by some managers to induce these production responses (see the section "Social Aspects of Conservation Practices"). The prescribed grazing system could result in lower cost, and that would potentially justify the practice. From society's perspective, grazing practices may also reduce fire danger, provide other habitat improvements that are valued, or allow integration and adoption of other management practices that add value.

A recent synthesis paper (Briske et al. 2008) summarized key findings from many different studies about the benefits of rotational grazing systems as compared to a continuous, season-long grazing strategy. The main conclusion drawn from the review was that "rotational grazing as a means to increase vegetation and animal production has been subjected to as rigorous a testing regime as any hypothesis in the rangeland profession, and it has been found to convey few if any, consistent [ecological] benefits over continuous grazing" (Briske et al. 2008, p. 11). As noted in the review, there has generally not been an economically measurable difference in plant production/standing crop or animal production between rotational and continuous grazing with similar stocking rates. The production function does not appear to shift up within a given year from improved animal performance or over time because of increasing forage production. Economically, this means that if rotational grazing requires more labor, capital, and management inputs, the lower-cost continuous grazing alternative would be preferred based solely on net ranch returns. This preference may be altered by other goals, such as nonvalued ecosystem services or other societal benefits.

Considering specifically the economics of implementing grazing systems, the CAB abstracts populated with 60 citations using the key words of "grazing systems" in the title and "economics" and "rangelands" in any other field. Further screening indicated only 23 of the articles were relevant. Nearly half the economic studies were conducted in Switzerland, Africa, and Australia. Some studies compared primarily different stocking rates or grazing intensities (Behnke 2000; Rook et al. 2004; Trapnell et al. 2006). Two African studies evaluated the

> "…economic evaluations of grazing systems have consistently found season-long continuous grazing to be the most economical or not different in production and rate of economic return."

“The most valuable forage is not necessarily of the highest quality; rather, it is available when few alternatives exist.”

economics of multipaddock systems and found that few paddocks rather than many paddocks were most economical (Beukes et al. 2002; Mentis 1991). Hart et al. (1993) noted that cross-fencing and water development were important to achieve uniform utilization of forage and for minimizing grazing energy costs, but these goals can be achieved independently of the grazing system.

Not surprising, given the finding of the Briske et al. (2008) literature review that few forage and livestock benefits accrue from rotational grazing, economic evaluations of grazing systems have consistently found season-long continuous grazing to be the most economical or not different in production and rate of economic return (Heitschmidt and Kothmann 1980; Quigley et al. 1984; Van Tassell and Conner 1986b; Hart et al. 1988a; Heitschmidt et al. 1990). A somewhat different conclusion was reached by Owensby et al. (2008) where season-long stocking of a tallgrass prairie site in Kansas was found to have the lowest economic risk (i.e., variability), but returns per hectare were higher for an intensive early stocking system. Given added capital and labor requirements for more intensive grazing systems, not only must there be measurable production responses from the practice, but those responses must be substantial enough to justify the added expense. The literature does not show this to be the case, and, as noted in the prescribed grazing chapter, there is minimal information documenting the influence of intensive grazing systems on the effectiveness of adaptive management.

Briske and coauthors (this volume) extensively explored published literature dealing with grazing/wildlife interactions. They note that many wildlife species, including birds and wild ungulates, demonstrate a relative neutral response to the type of grazing system in place. Both positive and negative responses were noted. Given the general lack of response, it is not surprising that we found no studies that explored the economics of wildlife interactions and grazing systems.

Another area where grazing systems and deferred grazing has been shown to be beneficial is as an adjustment mechanism to drought and seasonal forage shortages. As noted by Tanaka et al. (2007), the seasonality of forage use is an important consideration in ranch planning because the number of forage alternatives is limited during certain months of the year, and some forages and harvested feeds are considerably more expensive. The most valuable forage is not necessarily of the highest quality; rather, it is available when few other alternatives are. A grazing scheme that leaves residual forage for carryover and use during a future short-supply period, allows riparian areas to be rested, extends the grazing season, and/or replaces an expensive feed alternative has substantial economic value (Stillings et al. 2003; Tanaka et al. 2007).

Greater reliance on livestock grazing compared to harvested forages is an effective way to reduce feed costs that requires a planned grazing strategy. Adams et al. (1994) estimated that the weaning weights of calves were increased 5 kg by grazing meadows during May instead of feeding hay, and feed costs were substantially reduced. Extending the grazing season in winter and spring increases ranch returns over traditional systems that used a greater amount of harvested forage. Winter feeding costs are the largest expense for many livestock operations (Prevatt et al. 2001), and innovative grazing schemes have the potential to reduce those costs.

Briske et al. (2008) noted that even if evidence for production benefits from rotational grazing is inconclusive or nonexistent, many livestock producers believe that such benefits do exist. The website by Holistic Management International (http://www.holisticmanagement.com/n7/results_07.html; last accessed March 27, 2009) documented the stepped-up level of management and perceived benefits that ranch managers practicing holistic management have received. The survey of 43 ranch managers in the northern Rockies indicated that a high percentage of participants now do annual ranch planning, set goals, and have annual and formally documented land monitoring programs in place. It is from these activities that the majority of benefits from added management likely occur rather than a specific grazing practice. Further, they largely believed that production benefits do in fact exist, in contrast to the experimental evidence summarized by Briske et al. (2008).

Management and financial skills have typically been taught at holistic management schools, and these skills may be the biggest benefit that livestock producers have received from intensive grazing system training. These benefits are discussed in greater detail in the section "Social Aspects of Conservation Practices."

Brush Management

"The economics of brush control must be determined by the amount of forage and meat products gained; however, the principal objective in brush control should be an upgrade in range condition" (Hyder and Sneva 1956, p. 34). This statement, made over 50 yr ago, clearly articulated what was then and continues to be the main reason and economic rationale for brush control practices. The NRCS now recognizes six broad reasons for brush management (USDA-NRCS 2003):

- Added forage for livestock
- Restoration of natural plant community balance
- Creating the desired plant community
- Controlling erosion, reducing sediment, improving water quality, and enhancing stream flows
- Maintaining and enhancing wildlife habitat including protection of endangered species
- Protection of life and property from wildfire hazards

A Texas landowner survey describing incentive for brush control found that increased forage production and water conservation were most important (Kreuter et al. 2005). Secondary incentives were to improve aesthetic values, benefit the next generation, improve wildlife habitat, and improve real estate values.

Forage Production Benefits. The traditional brush control economic analysis that Hyder and Sneva (1956) described uses standard net present value (NPV) tools with the cost of the brush control treatment compared to discounted net future forage production benefits expected to be realized over some finite treatment life. The key tasks and elements of the economic assessment are to define the expected forage response (an assessment of forage productivity with and without the treatment) and estimate the added livestock carrying capacity possible over time with brush control, select an appropriate discount rate to properly account for timing difference between benefits and costs, and price and value the added grazing capacity (Workman and Tanaka 1991). A positive NPV or a benefit-to-cost ratio greater than one implies an economically feasible rangeland management practice (Workman 1986).

The economics of controlling brush for enhanced livestock production is variable depending on the economic value assigned to the forage, the assumed rate of forage response and treatment longevity, the assumed proper use rate or allowance for how much of the additional forage will be harvested to generate additional livestock income, and the discount rate used. The economic value of the added forage is influenced by the quality of the forage for grazing, by the forage and feed alternatives available, and by livestock prices. Most important is whether the added forage would be available during periods when other forages are scarce and costly (Evans and Workman 1994).

The expected longevity of brush control treatments varies widely by brush species and range site, as does the forage response. Yet a consistent overstory–understory relationship has been noted for many shrubland communities and species. These relationships generally show a downward-sloping sigmoid or exponential curves when herbaceous yield ($kg \cdot ha^{-1}$) is plotted against brush canopy (%; Ffolliott and Clary 1972). This suggests that increased brush cover diminishes understory forage production but at a decreasing rate.

Herbaceous production has been shown to increase an average of three to five times following effective control of many brush species located on productive range sites, including sites infested with Wyoming big sagebrush (*Artemisia tridentata wyomingensis*; Hyder and Sneva 1956; McDaniel et al. 2005), broom snakeweed (*Gutierrezia sarothrae*; McDaniel et al. 1993), pinyon-juniper (Clary et al. 1974; Pieper 1990), and redberry juniper (*Juniperus pinchotii*; Johnson et al. 1999). Successful control of other species like mesquite (*Prosopis glandulosa),* salt cedar (*Tamarix*

> “…forage production, not water yield, is the primary benefit of the brush control practice, especially on more mesic sites.”

spp.), and creosotebush (*Larrea tridentata)* resulted in an increase in grass cover, but with minimal changes in harvestable forage except on productive sites with adequate rainfall (Ethridge et al. 1984; Harms and Hiebert 2006; Perkins et al. 2006; Combs 2007).

The economics of brush management practices continues to be evaluated on the basis of the amount of forage and meat products gained by implementing the practice. The economic component of PESTMAN, a holistic decision support system currently in development at Texas A&M University (PESTMAN 2009), is driven by the anticipated forage response to a selected brush control treatment. Yet, as noted over 30 yr ago by Smith and Martin (1972), based on livestock production value, most rangeland management practices showed a negative benefit-to-cost ratio (costs exceed benefits) based only on the value of the added forage. This is a consistent and continuing conclusion from studies dealing with the economics of brush control practices. Increased returns from improved animal performance and production are usually too low for brush control to be economically justified (McBryde et al. 1984; Lee et al. 2001; Torell et al. 2005a). Landowners recognize this, and many brush control projects are implemented under cost-share arrangements with state and federal land management agencies.

Torell et al. (2005a) found that a cost-share payment of about 30% of the treatment cost was required to justify control of big sagebrush in northwestern New Mexico when the added forage from the brush control practice was valued at an intermediate level of $7 · animal unit month^{-1} (AUM; in 2003 dollars). The NPV of the investment was positive except at two relatively unproductive sites, when forage was valued at $10 · AUM^{-1}. A rangeland management practice that adds forage during a critical and limiting season makes forage valuable, and many times added forage production alone justifies the improvement in these cases (Evans and Workman 1994).

If a brush control project successfully increases available forage, increasing livestock numbers is not always justified. In some cases, a justification for brush control is that the stocking rate on the area can be maintained nearer its actual capacity by recognizing that current stocking rates are not sustainable. Sagebrush control near Farmington, New Mexico, helped the Bureau of Land Management (BLM) avoid potential conflict and lawsuits with grazing permittees and other parties because positive steps were taken to reduce grazing pressure but without forcing major herd reductions (Torell et al. 2005a). Similarly, forestalling the need for controversial grazing reductions was a primary benefit of the 11-yr (1962–1972) Vale Rangeland Rehabilitation Program initiated on BLM lands in eastern Oregon (Bartlett et al. 1988). Therefore, the maintenance of sustainable livestock carrying capacity should be given explicit consideration when evaluating the effectiveness of brush management programs.

Watershed Benefits. Watershed benefits are increasingly used as justification for public expenditures for brush control. Large tracts have been treated at great expense to control brush, especially salt cedar, to realize perceived watershed benefits, including added stream flow, water yield, and aquifer recharge. For various reasons, trees and brush are perceived to have a higher evapotranspiration (ET) rate than herbaceous species within the understory (Wilcox and Thurow 2006). The argument is made that if ET loss can be reduced by managing rangelands for a greater grass component and a lesser tree and shrub component, more water will be available for runoff and/or deep drainage. As noted by Wilcox and Thurow (2006), this argument has been shown to be true in a variety of humid, montane and Mediterranean climates, where studies have shown increases in water yields tied to removal of trees and shrubs. In semiarid rangelands, however, water yield benefits have not been demonstrated on scales that would greatly alter regional water supplies (Wilcox 2002). Sturges (1983) noted that the response of the soil water regime to a brush control treatment is inversely related to the response in herbaceous production. This suggests that much of the added water from brush control in arid areas is used to produce greater herbaceous production. Thus, added forage production, not water yield, is the primary benefit of the brush control practice, especially on more mesic sites. It appears that there is no real potential for increasing stream flows in addition

Prescribed burning mosaic, near Lakeview, Oregon. (Photo: John Tanaka)

to the forage benefit unless annual precipitation exceeds 450–500 mm (Wilcox 2002).

Research addressing the economics of brush control for enhanced water yield has been conducted only on Texas watersheds. As noted by Wilcox (2002), the perception is widespread that the water supply in Texas can be substantially increased through aggressive control of mesquite and juniper. Several studies have explored the cost implications of Texas brush control practices from a rancher perspective (Lee et al. 2001; Olenick et al. 2004a), and surveys have been conducted to evaluate ranchers' willingness to participate in brush control projects designed to enhance water yields (Thurow et al. 2000; Kreuter et al. 2004, 2005). These landowner surveys indicated that a subsidized public cost-share program would be necessary for widespread participation in brush control projects by Texas ranchers if public watershed benefits were the goal.

Economic studies have generally evaluated the economic value of watershed benefits indirectly using the procedure described by Lee et al. (2001) and Olenick et al. (2004a). The discounted forage production benefits are assumed to go to private ranchers, and they have an assumed willingness to participate at a maximum cost up to this level. Beyond this point, the NPV of the investment would be negative (benefits < costs) for the private landowner. Watershed benefits or the public's benefit is estimated to be the present value of the brush treatment cost minus landowner forage benefits. This residual value is a surrogate measure of the value that society must place on watershed benefits if the investment is to

Prescribed fire in pinyon-juniper, Arizona. (Photo: John Tanaka)

be economically efficient and have a positive overall NPV. The analysis makes no attempt to estimate what watershed benefits actually were but instead estimated the level of required public benefits required to justify the total cost of brush control. The obvious justification and assumption was that the cost-share program accurately reflected social priorities. Skeptics can counter, however, that land management agencies' budgets and spending priorities most often reflect political and bureaucratic objectives and are not a reflection of social value (Skaggs 2008).

Additional Texas studies have used plant growth, hydrologic, and economic models to determine costs and benefits (added water) resulting from brush control (Bach and Conner 1988; Conner and Bach 2000; Lemburg et al. 2002; Olenick et al. 2004a). Simulated estimates indicated that the public cost of additional water ranged from \$26 to \$129 per 1 000 m^3 depending on area and type of treatment. This compared with an estimated \$65 per 1 000 m^3 for leasing water pumped from the Edwards Aquifer (Olenick et al. 2004a).

Tamarix, or salt cedar, is often controlled on the basis of an economic justification associated primarily with water conservation. It is an expensive species to control (\$4 000–\$12 000 · ha^{-1}), requiring repeated mechanical, fire, chemical, and revegetation treatments as summarized at http://saltcedar.nmsu.edu. Economic feasibility studies assessing the costs and benefits of salt cedar control have been conducted for some western US waterways (Great Western Research Inc. 1989). Horton and Campbell (1974) estimated that water savings, based on the difference in water use between salt cedar and native vegetation, were as high as 3 000 acre-feet · yr^{-1} following salt cedar control on the Colorado River. It has been estimated that 568 000 acre-feet · yr^{-1} of water are lost to salt cedar from the Bonneville Unit of the Central Utah Water Project on the Colorado River at an estimated cost of \$27 million annually (Brotherson and Field 1987). Zavaleta (2000) estimated that marginal water losses to salt cedar are comparable to annual precipitation totals for the arid western states where salt cedar has been a problem. She estimated that *Tamarix* stands consume 3 000–

4 500 $m^3 \cdot ha^{-1} \cdot yr^{-1}$ (8 219–12 329 $L \cdot d^{-1}$) of water, more than the native vegetation it replaced. Lost economic value in 1998 dollars was estimated to be \$284–\$447 $\cdot ha^{-1}$ of land infested by the invasive species. Using a multiyear average treatment cost of \$5 000 $\cdot ha^{-1}$ and with a 6% discount rate, Zavaleta (2000) estimated that it would take 16 to 50 yr to break even on the initial and follow-up treatment costs, depending on the value assigned to the water saved.

The economic assessment by Zavaleta (2000) likely used the popular *Tamarix* water consumption value of about 757 $L \cdot d^{-1} \cdot plant^{-1}$. Owens and Moore (2007) reviewed the literature on water use by *Tamarix* and concluded that a more realistic estimate of the maximum daily water use was less than 122 $L \cdot d^{-1} \cdot plant^{-1}$. As Owens and Moore (2007) noted, at this reduced estimate of water savings, the economics of spending \$5 000–\$8 000 $\cdot ha^{-1}$ to control salt cedar would be dismal, based only on water conservation benefits.

Wildlife Benefits. Wildlife habitat and, therefore, wildlife populations are variously influenced by the relative cover of brush and tree species, and many wildlife species prefer a much denser overstory then would be optimal for livestock forage production and water yield alone (< 5% brush canopy). Surveys of Texas landowners indicated an average brush canopy cover of 41% on their ranches as compared to a preference of 27%, a level that would maximize the value of lease hunting for white-tailed deer (Thurow et al. 2000). Many Texas ranchers are more interested in brush thinning than brush eradication because of the high revenues derived from lease hunting (Kreuter et al. 2004).

Few studies have evaluated the economics of brush management for enhancing wildlife values and especially with consideration of the trade-offs with other resource values. Aldrich et al. (2005) developed a multiperiod linear programming model to evaluate the economics of western juniper control in central Oregon. Profit from livestock income was maximized by choosing economically optimal juniper management strategies so as to manipulate available forage resources for livestock production. Wildlife income was not considered a source of ranch revenue, but equations were included to evaluate how optimal juniper control strategies for livestock production would impact quail, deer, and elk numbers. Given the conflicting level of desired juniper for cattle versus wildlife production and with maximization of income from cattle only, wildlife numbers were projected to decline following livestock-revenue profit-maximizing strategies.

Standiford and Howitt (1993) developed an optimal control model that considered multiple resource values from management of California's hardwood rangelands, including forage production, oak wood sales, and hunting revenue. Key relationships were identified, including expected oak tree growth rates, the interaction of tree overstory versus livestock forage production, and hunting revenue potential under different tree canopies. Revenue from hunting was defined to increase with oak crown cover, whereas revenue from cattle decreased with an increasing canopy of oak trees. Economically, optimum oak canopy was found to vary depending on the resource values considered, but the optimum was consistent with expectations. When only livestock production value was considered, the optimal control model indicated that oak trees would be gradually cleared because of the resulting additional forage for livestock. Over the assumed 13-yr planning horizon, oak canopy would be reduced from 55% to less than 5%, but immediate tree clearing was not feasible given imposed realistic budget constraints. Adding firewood harvest to livestock income resulted in a light tree harvest for firewood (2–3% of oak canopy). The NPV addition from firewood revenue was 3%. Managing for cattle and wildlife increased NPV by 40% and greatly changed management of the oak canopy. The oak canopy would be maintained at 55%, no firewood would be harvested, and reduced cattle numbers would be optimal. The marginal economic value of the oak canopy for wildlife habitat exceeded the marginal value of greater livestock forage and firewood volume.

Bernardo et al. (1994) highlighted that wildlife is an increasingly important source of ranch income, and hedonic ranchland valuation

Spraying salt cedar along Pecos River. (Photo: Kirk McDaniel)

models show market preferences for ranches with wildlife revenue and hunting potential. Ranches with quality wildlife habitat and scenic appeal bring premium prices in the ranch real estate market. Torell et al. (2005b) estimated that 25% of New Mexico ranches have wildlife income-earning potential and that these ranches sell for premium prices, especially scenic mountain ranches with elk herds present.

As discussed above, Standiford and Howitt (1993) have shown optimal brush management strategies for wildlife to be different from optimal production strategies for livestock on California's hardwood rangelands. Bernardo et al. (1994) similarly concluded that some reduction in cattle grazing was necessary to maintain deer and quail habitat at desired levels. As noted by the Texas Parks and Wildlife Management Department (2008), most wildlife species are selective foragers, preferring to feed on a wide variety of plants rather than a few specific ones. Therefore, habitat improvement recommendations should emphasize the need for an even distribution and high availability of potential forage plants from season to season. Brush management can allow solid stands of woody vegetation to be interspersed with cleared areas over the landscape. Cleared strips or blocks can produce desirable forb and browse production while retaining an adequate mosaic of woody cover for escape, nesting, or protection from the elements. Properly utilized brush management practices can improve the availability of escape cover and food plants for both wildlife and livestock, though livestock production profitability will not be maximized.

Prescribed Burning

Various search engines, including AGRICOLA (Agriculture abstracts), CAB Abstracts, and wildlife and ecology studies worldwide, were used to evaluate the peer-reviewed literature that is available on the economics of prescribed fire as a conservation practice. "Fire" was required in the title and key words, or the title had to include "economics." After some screening and elimination of irrelevant papers, 46 citations dealt with fire and economics. Seven of these papers also used the term "social" in either the title or the key words. Six of the papers dealt primarily with fuel reduction on forested lands using prescribed fire.

Fire Hazard Reduction, Liability, and Risk Concerns. Of the papers dealing with human and economic aspects related to rangeland fires, nearly half the papers dealt with wildfires and using prescribed burning as a way to reduce the risk and economic damage from wildfires (Kaval et al. 2007; Mercer et al. 2007; Yoder and Blatner 2004). As noted by Kaval et al. (2007), residents within wildland–urban interface zones recognized the positive role that prescribed burning can have in reducing the dangers of wildfire and are willing to pay for positive fire risk mitigation measures. In this Colorado study, residents responding to the willingness-to-pay survey were willing to pay an average of nearly \$800 $year^{-1}$ for fuel management treatments to reduce fire risk. At a low \$5 · $household^{-1}$ annual bid price, all survey respondents were willing to pay for prescribed fire so as to reduce fire danger, while at a relatively high level of \$1 500 · $household^{-1}$ annually, only 14% were willing to pay for prescribed fire treatments.

Cost studies have shown prescribed fire to be a cost-effective fuel reduction method. Yet Hartsough et al. (2008) found that using prescribed fire to reduce fire danger was relatively high cost in the western United States because of terrain and stand conditions, high fuel loads, and the need to ensure that prescribed fires do not escape. Evaluating data

at seven sites in the western United States indicated that gross costs of mechanical fuel reduction treatments were more expensive than those of prescribed fire, but net costs were similar or less after the market value of the harvested wood products were deducted. The relative merit of using prescribed fire versus mechanical canopy removal was highly sensitive to the market value of forest products generated by the mechanical operation (Hartsough et al. 2008). As noted by Omi (2008), fire hazard reduction through fuels management is controversial, and the literature on fuel treatment effectiveness in reducing fire danger is nearly nonexistent, as are the data required to assess the trade-offs between alternative fuel reduction treatments.

Seven papers addressed fire escape and the liability risk associated with prescribed fires, all published since 2000. As noted by Yoder (2008), prescribed fire is considered a useful but risky method of reducing wildfire risk, increasing forage production, and improving wildlife habitat. This risk has resulted in new laws and reforms (Sun 2006), and Yoder found that these new liability laws and regulations have effectively reduced the incidence and severity of escaped fires.

Improve Forage Production. Some type of intervention may be needed to improve range condition and to redirect what may be a continued decline in rangeland productivity. However, Fuhlendorf and coauthors found a very weak argument for using fire to increase herbaceous vegetation production, particularly perennial grasses (this volume). They note that perennial grasses generally declined in the years immediately following prescribed fire but that most perennial species recovered within 2–3 yr. The delay in realized grazing benefit is problematic for realizing positive economic returns from the prescribed fire treatment given the time value of money. Further, conducting a prescribed burn may be limited or delayed by air temperature conditions, relative humidity, wind speeds, and the availability of fine fuels to carry the fire so as to minimize the risk of fire escape, to minimize damage to perennial grasses, and to carry the fire over the desired area. Fire is not as easy or as convenient to use as chemical treatments for brush control. McDaniel et al. (1997) noted that over a 6-yr study period on the shortgrass prairie of New Mexico, the desired fire conditions recommended by Wright and Bailey (1980) were rarely observed. For many arid rangelands, accumulating fine fuels under a dense brush canopy can be particularly problematic for implementing prescribed burning treatment (Bastian et al. 1995; McDaniel et al. 1997; Teague et al. 2001).

Eight papers studied the economics of prescribed fire as a strategy for reducing brush overstory and increasing the production of understory forage species. Two of these studies were outside the United States (Trollope 1978; Henkin et al. 1998). Most of the US studies evaluated the economics of prescribed burning for control of honey mesquite and cactus in the Texas Rolling Plains (Teague et al. 2001, 2008). Other economic studies addressed oak–hickory forests (Bernardo et al. 1992), eastern red cedar (*Juniperus virginiana*; Bernardo et al. 1988), Macartney rose (*Rosa bracterata* Wendl.; Garoian et al. 1984), and big sagebrush (Bastian et al. 1995).

Of the limited economic studies about prescribed fire to enhance forage production, a common prescription was an initial chemical treatment to reduce the brush overstory followed by prescribed fire treatments at 5–7-yr intervals as a maintenance treatment (Van Tassell and Conner 1986a; Teague et al. 2001). The economics of prescribed fire treatments were estimated to be better than chemical treatments even if the burn treatment was considered to be considerably less effective in overstory reduction and longevity (Teague et al. 2001). This is because, ignoring fire risk, the cost of prescribed burning was assumed to be much cheaper than chemical treatments in all the studies. Bastian et al. (1995) estimated that the cost of prescribed fire was only half that of chemical treatment, and Teague et al. (2001) estimated the cost of follow-up burn treatments to be only 10% of the initial \$56.81 · ha^{-1} chemical treatment. Perhaps the biggest limitation of the economic studies of brush control, including both chemical and fire, was the estimation of the forage production response curves. Forage response can be expected to be highly variable depending on soils, pretreatment brush cover, climatic conditions, and forage and brush

“Perhaps the biggest limitation of the economic studies of brush control, including both chemical and fire, was the lack of data to estimate forage production response curves.”

Snow geese at Bosque del Apache National Wildlife Refuge, New Mexico. (Photo: John Tanaka)

species. The economic studies were largely based on relatively short response studies (< 10 yr) and various unmeasured assumptions about the rate of brush reinvasion. Given the general nonlinearity of overstory–understory relationship, the economic studies concluded that a relatively dense stand of brush must be present initially to be economically feasible for control by either fire or chemical methods. None of the studies mentioned or considered the reduction in perennial grass cover in the immediate years following prescribed burning treatments as compared to chemical treatments.

Wildlife Benefits. We found only two studies investigating the economics of prescribed fire to improve big game habitat and increase wildlife income. A study by Gonzalez-Caban et al. (2003) (also published as Loomis et al. 2002) developed production functions relating deer harvest response to prescribed burning. Diminishing marginal benefits were noted. An additional 445 ha of prescribed burn increased deer harvest by 33 head, whereas the next 1 502 ha of prescribed burn increased deer harvest by eight head. When compared to the estimated \$519–\$593 · ha^{-1} cost of conducting prescribed burns, the economic value of the added deer harvest was only 3.4% of the total cost for the first 445 ha burned.

Teague et al. (2001) studied the economic response of honey mesquite control in the Rolling Plains of Texas from both herbicide and prescribed fire treatments. They noted that burning at a 5–7-yr interval improved wildlife habitat. Their economic evaluation of herbicide and prescribed fire treatments was most sensitive to realizing a wildlife income response. If treatment on any part of the ranch increased wildlife income, then the NPV of the brush control treatment was substantially increased.

Rangeland Planting

Rehabilitation of rangeland by seeding and planting began in the western United States in the late 1800s, and, according to Heady and Child (1999), more literature exists on range seeding than any other practice in range management. They also note that the environmental movement after 1970 demanded less seeding of rangeland with monoculture species and more with native species. Rehabilitation and prevention of erosion have replaced increased forage production as the primary objective for

seeding public lands. Rangeland seeding is also considered advantageous for managing weeds and cheatgrass (*Bromus tectorum*; Young and Clements 2009).

The Heady and Child (1999) textbook chapter (chapter 24) provides a wealth of information about ecological considerations about rangeland planting. Seeding guidelines they identify include the following:

- *Is the seeding needed?* Removal of the competitive brush overstory may be adequate. Seeding has the greatest potential for profitable returns when native vegetation does not exist.
- *Is the climate favorable?* Successful plantings are infrequent in areas receiving less than 250 mm of precipitation per year, but failure for areas with greater than 600 mm of rainfall are less frequent.
- *Is the habitat favorable?* Select seeding sites with the most herbaceous response potential.
- *What species should be planted?* Consider seasonal forage demands and the potential to replace expensive feeding alternatives. Recognize that a mixed diet is generally more desirable and often will produce greater livestock gains than a monoculture.
- *Manage the seeded area.* Provide grazing deferment after establishment and do not overgraze.

The steps required to analyze the economics of revegetation projects include incorporation of all these considerations as they relate to potential costs, benefits, and risks (Workman and Tanaka 1991).

Economic literature evaluating rangeland seeding are generally at least 30 yr old, and seeding success is variable and depends largely on selecting a desirable site in an adequate rainfall area. Much of the literature exists in the form of extension guides and bulletins. These bulletins generally provide guidelines for designing an economically successful project instead of evaluating the economics of specific improvement projects or case studies (Lloyd and Cook 1960; Wiens et al. 1969; Kearl and Cordingly 1975; Wambolt 1980; Kearl 1986; Workman and Tanaka 1991; Heady and Child 1999).

The probability of successful seeding establishment was highlighted in the only study found concerning the economics of range reseeding in the desert Southwest (Ethridge et al. 1997). This 6-yr study of seeding trials on the Jornada Experimental Range near Las Cruces, New Mexico, considered 14 different plant varieties, including introduced and native species. The study indicated that reseeding was not an advisable financial investment for the Chihuahuan deserts of southern New Mexico because of the high probability of stand failure. Estimated NPV was negative for all species planted and seedbed preparation strategies.

Most of the economic studies on rangeland planting—or rangeland seeding or reseeding, as it is categorized in the literature—are about the economics of seeding crested wheatgrass (*Agropyron desertorum* and *A. cristatum*). From 1945 until 1965, several million hectares of sagebrush rangeland were seeded to crested wheatgrass in the Intermountain West (Young 1994). It was estimated that in Nevada, 0.4 million of the 11 million ha of sagebrush rangeland were seeded to wheatgrass. The seeded area constitutes only 2% of the total rangeland in Nevada but produces 10% of the harvestable rangeland forage (Young and Evans 1986).

Seeding sagebrush rangelands to crested wheatgrass was generally found to be a very economical practice because forage production was 3–20 times greater than that of the native plants it replaced, calf crop and average weaning weights increased, and early spring use replaced expensive hay as an alternative lower-cost feed. In some studies, rates of return were estimated to be in the range of 10–22% with an anticipated stand life of 25 yr or more (Kearl and Cordingly 1975; Shane et al. 1983). Not all economic studies estimated positive economic returns, however. Godfrey (1986) reviewed 24 economic studies conducted between 1943 and 1979 that dealt with seeding crested wheatgrass and found net economic returns to be positive in nine of the studies and variable or unknown in the other studies. He attributed the variability in NPV estimates to four main reasons: 1) some of the plantings were failure, 2) low-production areas were seeded instead of high-potential areas (they used a worst first selection criteria), 3)

Bison at the Wichita Mountains Wildlife Refuge, Oklahoma. (Photo: John Tanaka)

too much was spent for brush removal, and 4) other, less productive improvements were included along with the seeding project.

Widespread planting of crested wheatgrass is no longer common, as it is an exotic monoculture, and successful widespread planting altered sagebrush habitat required for sage-grouse (*Centrocercus urophasianus*; Connelly and Schroeder 2000). As noted by Young and Evans (1986), the golden age of planting crested wheatgrass lasted for barely a decade, from the mid-1950s until the mid-1960s. Its role is now considered to be in the reclamation of drastically disturbed lands (Depuit 1986), and it has potential when seeded with forage kochia (*Kochia prostrata* ssp. *virescens*) to outcompete cheatgrass (*Bromus tectorum*; Harrison et al. 2000). Widespread use and plantings indicates that this forage species was one of the most economically successful rangeland plantings in its day, when only livestock forage production was valued.

Riparian Herbaceous Cover

Literature dealing with the economics of improving riparian herbaceous cover focuses primarily on nonrangeland (e.g., forested wetlands) areas and on tree cover rather than herbaceous cover. An EBSCO search with the terms "contingent valuation" and "riparian" returned 25 citations, but there were no studies found that specifically dealt with the economic value of establishing riparian herbaceous cover in rangeland areas as a way to improve riparian areas.

One of the few economic studies, a contingent valuation to estimate the benefits and costs of riparian restoration projects along the Little Tennessee River in North Carolina (Holmes et al. 2004), found net benefits from riparian ecosystem restoration to be strongly positive but much larger for large-scale projects. Restoration benefits were described in terms of five indicators of ecosystem services: abundance of game fish, water clarity, wildlife habitat,

allowable water uses, and ecosystem naturalness. Other economic studies explored the willingness to pay and willingness to accept payment for provision of a riparian strip or corridor for habitat preservation (Amigues et al. 2002; Qiu et al. 2006). The equivalent of a positive benefit-to-cost ratio was indicated with results consistent with housing price differentials (house prices with or without riparian strip or corridor for habitat preservation) for stream access in the area (Qiu et al. 2006). The willingness of recreational visitors to pay for riparian area preservation that maintained bird diversity in the San Pedro River basin of Arizona was explored by Colby and Orr (2005). They estimated that a one-time aggregate monetary willingness to pay by nonlocal visitors for riparian area preservation was $2.77 million.

Upland Wildlife Habitat Management

As previously noted, wildlife income is an increasingly important part of total ranch income, and ranchland market values are greatly influenced by scenic views, recreation, and hunting opportunities. Torell et al. (2005b) found that adding wildlife income to a New Mexico ranch contributes 2.5 times more to ranchland market value then does a similar amount of livestock income. These ranchland real estate market effects have been noted for many years (Pope 1985; Torell et al. 2005b). The obvious implication is that significant opportunities exist to increase economic values through upland wildlife habitat management if increased hunting and wildlife viewing opportunities can be created.

A great deal of research has been conducted to estimate the demand and economic value of various wildlife species, with obvious value implications for wildlife habitat improvement. One notable author, John B. Loomis at Colorado State University, has contributed greatly to the development and application of nonmarket valuation procedures. However, much of his research and other related research, as noted by Daniels and Riggs (1988), has concentrated on estimating the value of the wildlife and not the habitat sustaining wildlife. As noted by Bernardo et al. (1994), the weak link is the lack of data required to translate physical effects of habitat improvement practices into altered wildlife numbers and economic benefits. The Bernardo et al. paper provides one of the limited cases where the production trade-offs between cattle grazing and wildlife habitat were estimated. A second study was a California study on oak rangeland described earlier by Standiford and Howitt (1993). A third study involved estimation of the production possibility frontier between stocker cattle and antelope by Bastian et al. (1991). Glover and Conner (1988) developed a linear programming model to evaluate the optimal (profit-maximizing) mix of cattle, sheep, goats, and deer on a representative ranch in the Edwards Plateau region of Texas and found that active management for wildlife added to net ranch income. Another study by Loomis et al. (1991) evaluated livestock grazing strategies that would potentially improve deer habitat in California and concluded that implementing a rest–rotation livestock grazing system with 1 yr or more of nonuse in a 3-yr cycle would increase hunting value far beyond the value lost from reduced livestock grazing. Others have estimated forage values for cattle versus wildlife (Martin et al. 1978; Cory and Martin 1985; Loomis et al. 1989), but they did not provide estimates within a multiple-enterprise context that estimated production possibilities and trade-offs.

Wildlife valuation procedures use various techniques to estimate a consumer's willingness to pay where no established market exists, relying on demand analysis and consumer surplus estimation (Sorg and Loomis 1985; Champ et al. 2003). Valuation of wildlife habitat uses these value estimates for wildlife and expands to a benefit-to-cost assessment where the economic value of increased wildlife numbers is compared to the cost of practices that improve habitat and ultimately wildlife numbers. Studies that have attempted to estimate the linkage between altered habitat and wildlife numbers include the prescribed burning assessment described above (Loomis et al. 2002; Gonzalez-Caban et al. 2003) where benefits from additional deer harvest was estimated to be no more than 3.4% of prescribed burning treatment cost, suggesting that deer hunting benefits represent only a small part of the multiple-use benefits of prescribed fire.

Lenarz (1987) concluded that treatments to increase forest openings were never cost

> "Very little if any research exists showing the direct noneconomic effects of NRCS rangeland conservation practices on individuals, households, or social systems."

Planning in the tallgrass prairie, near Bowie, Texas. (Photo: John Tanaka)

effective based on the value of hunting licenses, but it was cost effective based on total gross hunting-related expenditures. Daniels and Riggs (1988) recognized and corrected the low economic value assigned by Lenarz (1987) by considering only the value of the hunting leases. They based deer values on standard willingness-to-pay measures and concluded that a positive NPV would be realized from investments to create forest openings whenever cleared areas were less than the 3% level and a reasonable discount rate was used.

Garrett et al. (1970) estimated the demand for deer hunting in Nevada and valued the habitat that supported hunting activity. They compared the habitat value to selected rehabilitation projects expected to alter deer numbers. The first improvement considered was crested wheatgrass (*Agropyron cristatum*) seeding with the estimate that the practice would be detrimental to deer numbers and wildlife value because of the decrease in forage species most desirable to deer. Chaining of pinyon-juniper at two sites was estimated to result in a positive benefit-to-cost ratio, ranging from 1.65 to 2.09 for the two sites, based only on increased economic value from deer utilization supported by a more open woodland canopy.

Similar to the hedonic modeling approach used by Torell et al. (2005b) to evaluate the contribution of wildlife to western ranchland values, Netusil (2006) used urban housing sales within the Fanno Creek Watershed within the city of Portland, Oregon, to evaluate how real estate prices varied with different amounts of upland wildlife habitat. Close proximity to a stream increased property values. A property's sale price was found to increase as the percentage of regionally significant habitat on the lot increased but at a decreasing rate. Property owners placed a premium price on lots with habitat providing the highest ecological values (large forest patches, wetland areas, and large contiguous patches) and

discounted lots with lower-valued habitat. The maximum impact on house lot sale price was when upland wildlife habitat coverage on the property was about 38%.

SOCIAL ASPECTS OF CONSERVATION PRACTICES

Very little if any research exists showing the direct noneconomic effects of NRCS rangeland conservation practices on individuals, households, or social systems. It is likely that many producers do realize psychological benefits from conservation, as stewardship outcomes typically rank high among the management goals of livestock producers (Huntsinger and Fortmann 1990; Sayre 2004). Moreover, livestock producers who believe strongly in a responsibility to society are more likely to engage in environmentally desirable management practices, such as invasive weed control and riparian protection (Kreuter et al. 2006). Thus, non–peer-reviewed feature articles often refer to the psychological rewards ranchers enjoy because they employ conservation practices on land they hope to preserve for posterity (e.g., Little 2005; Smith 2008). Such rewards are often hard to document scientifically, however.

Indirect evidence of psychological benefit from implementing conservation practices comes from Holistic Management (HM), a program that typically advocates rotational grazing as part of an overall ranch management plan. Montagne and Orchard (2000) found that participating ranchers in the northern Rockies reported increased personal satisfaction after having adopted an HM approach. Stinner et al. (1997) found after interviewing HM participants nationwide that 91% reported improvements in quality of life after HM training. However, HM is a whole-ranch program that focuses on time management, goal setting, and monitoring as well as prescribed grazing, and neither study separated the effects of different aspects of the program. Moreover, such studies can speak only to the *perceived* effects of conservation; we have found no evidence that land managers, farm/ranch households, or group members that engage in conservation practices actually score higher on measurements of psychological or social well-being than producers who do not use such practices.

Much more is known about why people choose to adopt conservation practices than the relative effectiveness of their implementation. Studies of innovation adoption offer insight as to which outcomes are anticipated by landowners and managers and the circumstances under which those outcomes are likely to be sufficiently valued to produce a change in management practice. Several thorough reviews

New residential subdivisions in former ranchland, Eagle Mountain, Utah. (Photo: Mark Brunson)

have been produced over the years (e.g., Nowak and Korsching 1983; Clearfield and Osgood 1986). Most recently, Prokopy et al. (2008) reviewed 55 separate studies over a 25-yr period that explored adoption of agricultural best management practices (BMPs) in the United States. Their goal was to identify general trends in how adoption of conservation practices is related to social-psychological, enterprise-based, and social and economic factors. Most of the reviewed studies focus on soil, nutrients, and pest management; very few focused on the water or livestock management practices pertinent to grazing lands. Nonetheless, their findings offer general guidance about the role of anticipated benefits in the implementation of practices.

The variables most strongly associated with adoption of BMPs were attributes of the decision maker or of the farm/ranch operation: demographic factors, such as income and education; access to information, capital, and social support; and farm size (Prokopy et al. 2008). Producers' awareness of environmental problems and their overall environmental attitudes were positively associated with BMP adoption, not surprisingly suggesting that farmers and ranchers are more likely to adopt conservation practices if they believe that conservation is important.

Rogers (2003), whose theories on innovation adoption and diffusion have been highly influential in many fields including agriculture, identified three categories of factors that affect adoption rates: attributes of the potential adopter, the adopter's social system, and the innovation itself. In this chapter, we are most interested in the latter category, as it encompasses the perceived personal, family, or social benefits that potential adopters ascribe to conservation activities on grazing lands. Wejnert (2002) further divides the pertinent innovation attributes into two decision metrics: the degree to which the benefits of a practice are thought to outweigh the costs and the degree to which the positive or negative consequences of adoption accrue to the private individual versus the public good. Both criteria relate to producers' beliefs about personal, social, and economic factors as well as environmental benefits of a practice under consideration for adoption. Perceived costs can be psychological or social, just as are benefits; for example, Grigsby (1980) argued that one of the most significant barriers to innovation among ranchers is a belief that the innovation somehow threatens their ranching lifestyle.

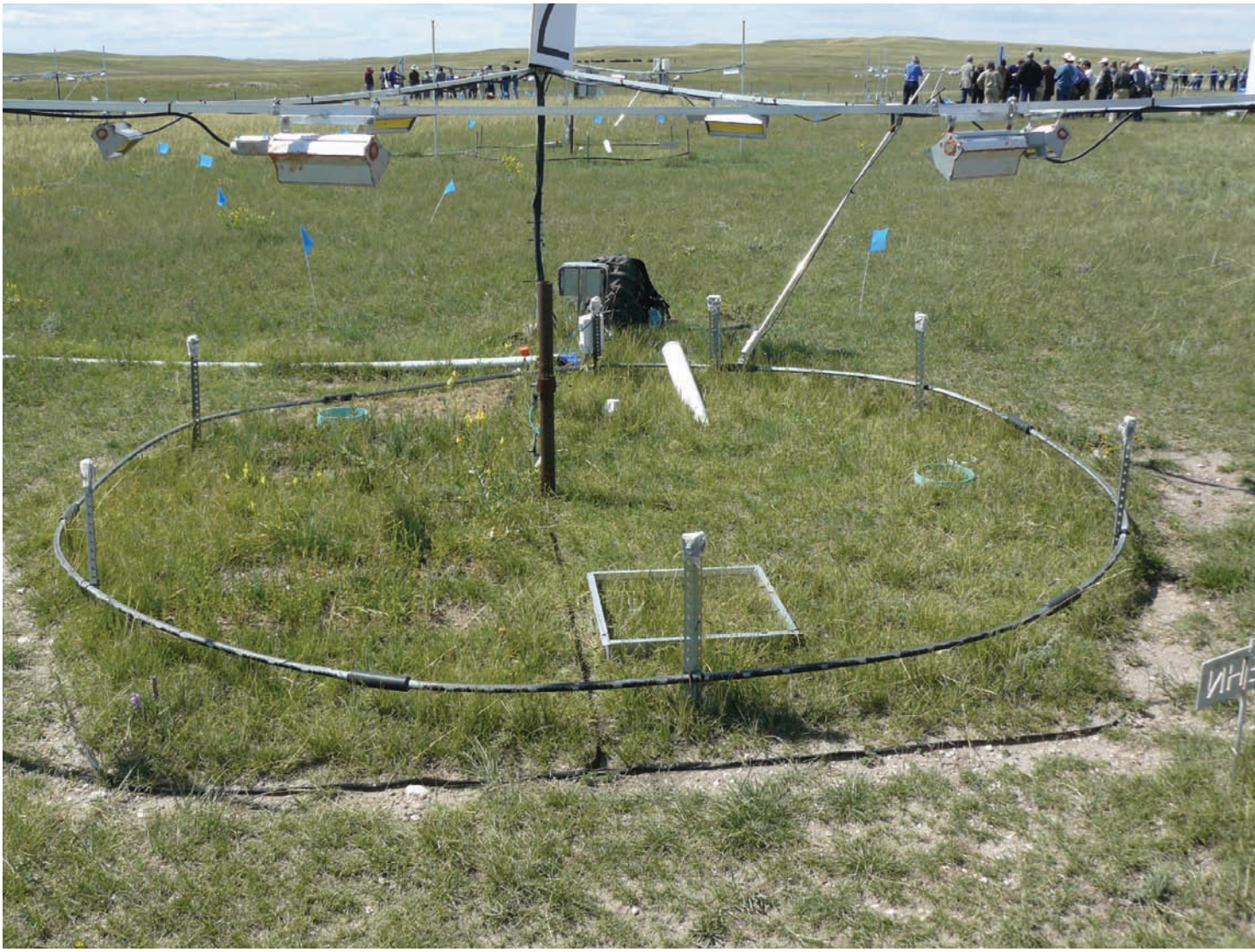
INNOVATION–ADOPTION STUDIES

Innovation–adoption studies abound in agriculture. Most of these focus on crop producers, but a number of researchers have explored practices recommended by the USDA-NRCS for grazing lands conservation. In this section, we describe more general studies that may include multiple practices; practice-specific research is described in subsequent sections.

In southwestern Oregon, Habron (2004) found that landowners implemented upland *conservation practices* such as off-stream *livestock* water developments and rotational grazing more often than fencing or tree planting in riparian areas. Key influences on whether producers adopted any practice at all were whether they used irrigation, shared management decisions with a spouse, believed in scientific experimentation, and discussed *conservation* with others. The key factors predicting adoption of specific BMPs depended on the kind of *practice* implemented.

A Utah study asked ranchers who have reputations as innovators what outcomes had led to their adoption decisions (Didier and Brunson 2004). One of the most influential outcomes was social: interviewees often reported that they were motivated by a desire to demonstrate stewardship to federal land managers and/or the public. The authors did find that innovation attributes were important to ranchers, especially in the negative. For example, interviewees reported that they or neighbors had rejected conservation practices because of perceived poor cost-to-benefit ratios or difficulty in pilot-testing a practice before full adoption. Brush management was cited as an example of the former barrier, while prescribed grazing—especially in the form of a short-duration rotational grazing system—was typically noted as an example of the latter.

Barao (1992) surveyed Maryland producers who had attended an extension field day



Measuring CO_2 and temperature effects, High Plains Research Station, Cheyenne, Wyoming. (Photo: John Tanaka)

to learn which demonstrated practices, if any, were subsequently adopted. Grazing management was the most common practice adopted (34%); livestock nutrition improvements, such as changing pasture species composition or analyzing forage/feed, were less commonly adopted. Perceived outcomes of the practices were not found to be as important to these decisions, however, as the ease with which the practice can be learned and the results of a change can be observed. In another assessment of an extension program, a Livestock Systems Environmental Assessment tool, Koelsch et al. (2000) found that producers who used the tool were most likely to cite a desire for improved environmental stewardship as the most important reason for doing so.

Nonadoption

A few studies have taken the opposite approach to understanding adoption decisions, and these also may prove useful—if people are not adopting conservation practices because they do not consider them worthwhile, that would suggest that such practices are not thought to provide personal benefits. However, Gillespie et al. (2007) found in a survey of 1 700 Louisiana beef producers that the most influential reasons for nonadoption of 16 BMPs were because they felt the practice would not work on their property or were unaware of the practice. Similarly, Prokopy et al. (2008) found that access to information influenced likelihood of adoption. Thus, knowledge of USDA conservation programs can also be an impediment to adoption. In

Kansas, Smith et al. (2007) found that 80% of survey respondents knew about EQIP; 31% participated in the program. Those figures are considerably higher than in an earlier study by Cable et al. (1999), who found that only 54% of survey respondents were aware of state or federal cost-share programs for private land conservation practices. For those who are aware of these NRCS cost-sharing programs but do not participate in them, lack of interest in conservation was not a significant influence on nonadoption; instead, ranchers tended to cite perceived regulatory impacts and paperwork as important reasons not to enroll (e.g., Didier and Brunson 2004; Smith et al. 2007).

Geographical Regions

Most of the studies described here pertain to a specific region, such as brush management in Texas (Taylor 2005; Kreuter et al. 2008) or prescribed burning in the desert Southwest (Sayre 2005). Some results of these studies are likely to be applicable to grazing lands nationwide, especially those relating to characteristics of land managers themselves, such as the importance placed on conservation as a management goal or sociodemographic and information access influences on adoption.

However, geographic factors are likely to influence landowners' beliefs about the perceived outcomes of conservation practices and thus the likelihood that those practices will be implemented. For example, Regen et al. (2008) found that protecting wildlife habitat and prairie restoration were important issues for a majority of landowners in a region straddling the Iowa–Missouri border. Most respondents reported using brush management practices to control eastern red cedar, but only 25% used prescribed burning, a practice frequently recommended for control of this species. In contrast, 38% of respondents to a survey by Kreuter et al. (2008) had used prescribed burning, mainly to control brush. Similarly, Liffman et al. (2000) found that 25% of landowners in Alameda and Contra Costa counties, California, had used prescribed burning in the previous 5 yr, while 34% of those in Tehama County had done so. In both the Midwest and California situations, a likely explanation for lower burning rates is likely to be the juxtaposition of grazing lands with other land uses (cropland in Iowa and Missouri and rapid exurban development in Alameda and Contra Costa counties), whereas burning is less likely to pose liability and permitting difficulties in areas where grazing lands dominate.

Prescribed Grazing

As was noted previously, researchers studying HM have found that practitioners report improved quality of life as a result of participation in their decision-making program (Stinner et al. 1997; Montagne and Orchard 2000). A similar conclusion was derived from the Sustainable Grazing Systems program in southern Australia. This program was established in 1996 to address declining pasture productivity. Nearly 10 000 Australian livestock producers received training and new skills, participated in demonstrations, and integrated management and goal setting into their ranching operations, similar to those participating in the HM program.

It is not known whether perceived improvements from program participation was a result of having employed a prescribed grazing system or some other factor associated with the learning experience but in the Montagne and Orchard (2000) study, interest in alternative grazing systems was one of the most frequently cited reasons for change. Overall, ranchers reveal positive ecological changes on the land and increased economic as well as personal satisfaction. In Minnesota, a psychological benefit was reported in collaborative research project by farmers and scientists that found that rotational *grazing not only* improved soil, pasture, and stream quality but also boosted the confidence of the farmers in their ability to employ more sustainable *grazing* practices (Badgley 2003). Again, it is not clear whether it was the grazing system or the collaborative process that improved farmers' confidence levels.

Collective interests of groups can also benefit from conservation practices beyond the individual benefits. Armstrong and Warner (1992) reported that adoption of a rotational grazing strategy by the Walker River Tribe in Nevada promoted tribal interests by benefiting all natural resources. This assumes, however, that resource benefits truly exist. As noted above, Briske et al. (2008) states that

Elk in the Pequop Range in Elko County, Nevada. (Photo: Tim Torell)

even if evidence for benefits from rotational grazing is inconclusive, many livestock producers believe that such benefit exists. They suggest that "personal goals and values . . . are inextricably integrated within grazing systems, and they are likely to interact with the adoption and operation of grazing systems to an equal or greater extent than the underlying ecological processes" (p. 10). A basic theory of psychology holds that people are motivated to behave in ways that are consistent with their beliefs, and when evidence suggests that those behaviors are unhelpful, they may tend to reject the evidence rather than reject the belief (Festinger 1957). Consumer researchers (e.g., Mano and Oliver 1993) have applied this theory to explain postadoption satisfaction levels, suggesting that the act of having adopted a new product or behavior predisposes one to evaluate it positively. If we apply this to rotational grazing, one explanation for the continued perception of realized benefits is that there are psychological rewards associated with doing so.

Moving cattle in the Blue Mountains, Oregon. (Photo: John Tanaka)

Brush Management

While brush management is a recognized NRCS conservation practice, it may or may not be considered "conservation," depending on the purpose of the practice and the historic and current conditions where it is implemented. For example, removing encroaching junipers to improve wildlife habitat and water availability in central Texas might be considered conservation, whereas removing all sagebrush from a native shrub–steppe community and planting a nonnative forage grass would not. Even in the former case, brush removal might constitute "conservation" if intended to benefit black-capped vireo (*Vireo atricapilla*) yet detrimental if the site is occupied by golden-cheeked warblers (*Dendroica chrysoparia*). Unfortunately, brush management implementation studies do not necessarily distinguish between management for conservation and management for other purposes.

Kreuter et al. (2001) surveyed Texas county extension agents to assess landowner interest in and adoption of Brush Busters, a collaborative extension/research program. Respondents reported that the landowners with whom they work perceive the program to be an "inexpensive, convenient, safe, effective, and predictable method for controlling brush." Interestingly, the authors recommended that to increase adoption rates, technology transfer professionals should emphasize the short-term economic benefits of Brush Busters rather than the long-term environmental benefits. This suggests that ranchers who implement this practice may not obtain personal benefits from implementing brush management as a conservation practice; rather, the benefits may accrue to society. Similarly, Thurow et al. (2001) found that economic factors were associated with ranchers' willingness to enter into a brush control cost-share contract but that conservation motives were not. Further evidence that conservation alone cannot motivate brush management comes from Olenick et al. (2005), who found in a 2003 Texas survey that landowners generally held favorable views toward programs that would reduce brush cover to increase water yields or to improve wildlife habitat, but they disapproved of programs that would encourage the proliferation of woody plants in an attempt to increase atmospheric carbon sequestration. Landowner attitudes were also associated with the voluntariness and flexibility associated with any proposed program to enhance ecosystem services.

Prescribed Burning

Prescribed burning is a practice where the conservation benefits are offset by potential

risks, including a loss of forage, regulatory difficulties associated with smoke and burning permits, weak legal protections against liability, and potential escape of the fire onto a neighboring property (Liffman et al. 2000; Brunson and Evans 2005). Ranchers may believe that prescribed fire would benefit their land but are reluctant to implement. For example, Sayre (2005) studied eight locations in southern Arizona and New Mexico where prescribed burning was part of wildlife conservation efforts on grazing lands. He found that while interest in restoring fire to the landscape was high, use of prescribed burning was limited by trade-offs between conservation goals and forage availability and by real or perceived regulatory scrutiny. Burning was most feasible where there were institutional structures that allowed for collaborative management across ownership boundaries.

In the Great Plains, an institution has arisen for just that purpose. Prescribed burning cooperatives offer landowners a chance to learn from peers how to apply fire safely and effectively and reduce liability concerns (Taylor 2005). In Texas, Kreuter et al. (2008) found that members of a large cooperative had more positive attitudes than nonmembers about the ecological role of fire and the use of prescribed fire. The authors suggest that the group not only offers opportunities to learn and reduce liability but also promotes cooperative behavior that can benefit a ranching community. Thus, in the context of a prescribed burning association, there may be a social benefit to implementing this conservation practice.

Rangeland Planting

Few authors have specifically addressed the social aspects of rangeland planting as a conservation practice, but these provide interesting insights as to the influence of changing societal norms. As noted above, seeding of crested wheatgrass was one of the most common vegetation management practices in the western United States during the 1950s and 1960s. By many estimates, it was also one of the most economical practices because of the species' competitiveness, early grazing use, and productivity. However, as the use and restoration of native plants has grown more popular, societal opinion has turned against the idea of replacing native rangeland with a monoculture of an exotic species that can persist for decades (Johnson 1986; Conner and Bach 2008). Negative characterizations of this highly adaptable and productive forage species have some critics who see planting crested wheatgrass as an ill-advised subsidy of ranching on public lands (Abbey 1988; Hess 1992) and those who complain that it has reduced sagebrush habitat for Greater sage-grouse (Connelly and Schroeder 2000). While it is still used on private ranches, crested wheatgrass seeding in public land grazing allotments has declined (Conner and Bach 2008), often restricted to areas vulnerable to invasion by exotic annual grasses where rapid revegetation is needed for site stabilization after wildfire. Even then, plans may call for the use of "assisted succession" (Cox and Anderson 2004) to replace crested wheatgrass with native perennial species as soon as is practical.

Young and Clements (2009, p. 179) described these largely social pressures affected rangeland management:

The politics of bureaucratic survival called for saying and doing as little as possible. Public land managers learned never to propose a seeding to increase forage supplies because government agencies, environmentalists, and archaeologists would descend en masse demanding a full environmental impact statement. Even wildfire burns were not seeded. When public pressure dictated seeding some very important habitat, the seed mixture was composed of species that had no chance of establishment and was seeded by aerial broadcasting on unprepared seedbeds. Young managers who tried seeding and failed were careful never to try again.

Riparian Herbaceous Cover

Considerable research has explored attitudes toward riparian restoration and protection, but almost all has focused on tree cover rather than herbaceous cover. For example, Lucht (2007) analyzed interest in adoption of agroforestry and conservation practices, including riparian planting, among agricultural producers, resident nonfarm landowners, and absentee nonfarm landowners. She found comparatively high interest but low knowledge levels compared to other practices. Absentee nonfarm landowners had the highest level of interest.

Public information campaign in an area undergoing transition from farm/ranch land to small-acreage subdivisions, Arimo, Idaho. (Photo: Mark Brunson)

Ryan et al. (2003) found that Michigan farmers were motivated primarily to adopt conservation practices along riparian zones, not for the economic returns it would provide but because of their strong attachment to the land and their desire to convey the message that they are good stewards of the land. They noted that strategies for conservation must respect farmers' attachment to the land, the desire to practice good stewardship while deriving income from the land.

One study that did focus on nonwoodland planting was by Smith et al. (2007), and they reached conclusions similar to Ryan et al. (2003) about developing strategies for conservation. They found that Kansas farmers and ranchers were less likely to plant riparian filter (buffer) strips than to employ other forms of best management practices because filter strips must be enrolled into the Continuous Conservation Reserve Program, thereby incurring restrictions on haying and grazing use.

Upland Wildlife Habitat Management

Upland wildlife habitat management, more so than other practices described here, is likely to enhance the use value for landowners as well as the bequest or existence values. The benefits that can be realized from wildlife are many and well known (Manfredo 2008). Because habitat enhancement also enhances the likelihood of being able to successfully view or hunt wildlife, many landowners will improve their land for wildlife. For example, when Cable et al. (1999) surveyed 900 Kansas agricultural producers about wildlife and riparian

areas, they found that more than a third of respondents reported that they had idled land or changed management practices specifically to help wildlife. The most common "extremely important" motivations for doing so were to preserve wildlife for future generations (55.6%) and because the landowner enjoyed watching wildlife (51.7%). A nationwide survey of farmers and ranchers by Conover (1998) found that about half of respondents manage their properties to enhance attractiveness to wildlife.

Both personal and monetary benefits influence decisions to implement wildlife habitat improvements, and it can be difficult to separate the two. Van Kooten and Schmitz (1992) found that agricultural producers participating in a waterfowl habitat enhancement project in western Canada held more positive attitudes toward wildlife than nonparticipants and therefore can be assumed to obtain personal benefits from participation, but positive attitudes alone were not sufficient to motivate habitat improvements in the absence of economic incentives. While this study was not completed in the United States, other studies conducted in this country (e.g., Troy et al. 2005) support the idea that the social or psychological benefits of wildlife habitat enhancement typically do not offset costs of doing so without some sort of economic incentive.

Incentive to entice landowners to adopt wildlife management programs may take several forms. Conover (1998) reported that nearly 80% of farmers and ranchers had encountered some sort of wildlife damage on their properties. This is one of several impediments to implementation of wildlife habitat improvements. Cable (2002) reported that while Kansas agricultural producers believe that it is important to protect wildlife habitat, fewer than half had set aside any of their property for wildlife. The primary reasons were fear of increased trespassing by hunters if their land became known as especially attractive to wildlife and the cost of idling any of their land. Costs are even greater when the wildlife may be protected under the federal Threatened and Endangered Species list (Brook et al. 2003; Elmore et al. 2007). Thus, any positive personal outcomes associated with implementing the upland wildlife habitat management practice must be great enough that a landowner is willing to take on the risk of other consequences, such as damage, trespass, or increased regulatory scrutiny and reduced management flexibility.

Incentives for wildlife habitat management on private land can be nonmonetary as well as monetary. Programs such as the Safe Harbor and Candidate Conservation Agreement with Assurances programs of the U.S. Fish and Wildlife Service as well as other collaborative land management efforts that exist across the West seek to protect landowners against regulatory risks in exchange for taking actions on behalf of wildlife (Belton 2008; Womack 2008). Belton (2008) surveyed members of local working groups that attempt to maintain habitat for the greater sage-grouse. Participation in these efforts can promote cohesion among landowner neighbors and enhance cooperation with government agencies, but the "peace of mind" that such efforts are intended to provide are dependent on trust levels in the agencies responsible for wildlife management and protection. As before, Belton's (2008) research suggested that people are more likely to participate if they receive monetary compensation for providing habitat for grouse.

VALUE OF ECOSYSTEM SERVICES

Types of Values

Ecosystem services benefit society in numerous and diverse ways. We can differentiate between those goods and services that are place bound (in situ) and those that can be derived from multiple locations (ex situ). There are a variety of classifications, including consumptive and nonconsumptive, market and nonmarket, primary and secondary, and in situ and ex situ (Brown et al. 2007; Cooper and Dobson 2007; Breckenridge et al. 2008). The basic issue is how to account for all the benefits and costs associated with the services derived from rangeland ecosystems. As noted earlier, each of the conservation practices can potentially produce different kinds, qualities, and amounts of these goods and services, depending on location, natural potentials, current states, and other factors.

Brown et al. (2007) used a traditional approach by dividing ecosystem goods into

nonrenewable and renewable. Nonrenewable goods included rocks, minerals, and fossil fuels, while renewable goods include wildlife and fish, plants, water, air, soils, recreation, and aesthetics. Ecosystem services include purification of air and water, nutrient cycling, maintenance and renewal of soil and soil fertility, pollination of crops and natural vegetation, dispersal of seeds, maintenance of regional precipitation patterns, erosion control, biodiversity maintenance, control of pests affecting plants or animals, protection from the sun's harmful UV rays, partial stabilization of climate, moderation of temperature extremes and the force of winds and waves, and mitigation of floods and droughts. In a listing of primary and secondary benefits from using pesticides, Cooper and Dobson (2007) divided primary benefits into agricultural production, energy needs, and preventing problems and secondary benefits into farming communities, national issues, and global issues. The interaction among all these benefit categories is complex and makes such separation difficult (see the section "Ecosystem Services").

Economic Valuation

In valuation of nonmarket ecosystem goods and services, there are few acceptable methods used in the literature. It is important to note that all these seek to estimate what a person or household would willingly pay to have that good or service or to put a value on damages from losses or costs avoided (Olewiler 2004). The comparability of these values with goods and services that are actually paid for out of the individual's income remains a question. In other cases, for the experiments to have validity, the consumer being questioned needs to have a very clear idea of what goods or services are at stake.

Before delving into the valuation question, it is important to note that the nature of ecosystem goods and services that are not traded in private markets lead to what is known as market failure and results in goods and services being either undersupplied or overused (Lant et al. 2008). Goods and services that are provided through private markets involve feedbacks that can potentially provide efficient levels of production. Rules or incentives put in place to deal with market failures can lead to inefficient levels of production and essentially require the regulating agency to guess at market-clearing prices or quantities. It is important that economists work closely with ecologists and other specialists in a truly collaborative process in their efforts to estimate values (Heal and Barbier 2006).

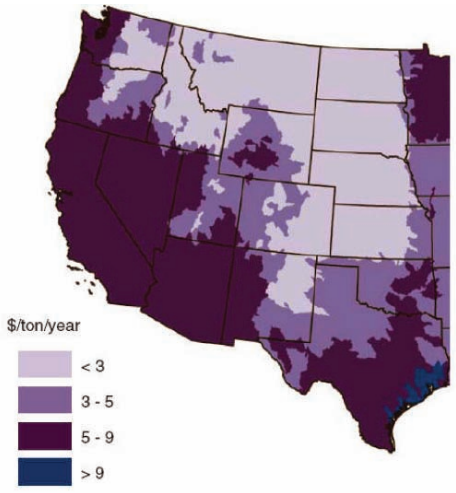
In a study that estimated the value of "diversity in biodiversity," Christie et al. (2006) found that the public did not generally understand different attributes of biodiversity even as they valued biodiversity itself. The public was also found to be relatively indifferent as to how biodiversity was achieved, but most attributes of biodiversity examined had positive values. Of course, biodiversity as an attribute is a multifaceted concept that occurs at many scales (West 1993), and one has to carefully define what is meant by the term before it can be valued.

It is also important to recognize who is likely to be the recipient of the benefits or who is setting the value (Burger et al. 2008). As they note, when valuing an individual species that does not have immediate or direct value, it is usually conservationists or regulators that set the value. On the other hand, when the individual species has a direct value to individuals, values are set through businesses, social scientists, or others with a direct connection to that species. The other types of resources they examined were the value of ecosystems to human communities and intact ecosystems with ecological, aesthetic, and existence values to people. Burger et al. (2008) also suggests specific economic value estimation methods that are appropriate for each type of environmental good or service (Table 5). The valuation methods that may be used in different situations are travel cost,

TABLE 5. Economic estimation methods suggested by Burger et al. (2008).

Type of ecosystem good	Methods
Resources themselves	Use survey of selected businesses
Specific resources for individuals	Use sample surveys to estimate direct values; estimate direct, indirect, and induced values using regional economic models
Resources for communities	Estimate replacement value, insurance costs, regional economics
Intact ecosystems	Use contingent valuation to estimate existence values; estimate regional economics

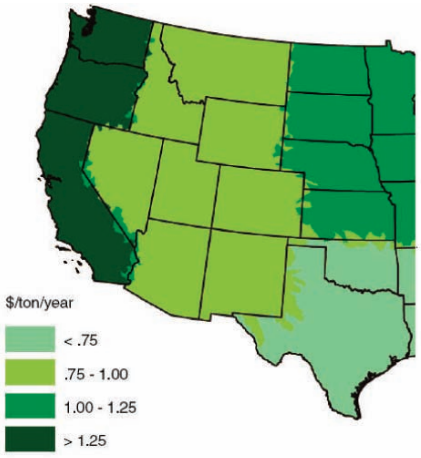
FIGURE 1. Water erosion benefit estimates (adapted from Hansen and Ribaudo 2008).



HUC-level[1] water-erosion benefit categories are: *reservoir services, navigation, water-based recreation, marine fisheries, freshwater fisheries, municipal industrial, steam electric, irrigation ditches, flood damages, soil productivity, road ditches*, and *municipal water treatment*. Only *reservoir services, navigation*, and *water-based recreation* are estimated at the HUC level. Other values are based on FPR-level[2] estimates.

[1]HUCs are watersheds defined by the U.S. Geological Survey's 8-digit hydrologic unit codes.
[2]FPRs are USDA's multi-state Farm Production Regions.

FIGURE 2. Wind erosion benefit estimates (adapted from Hansen and Ribaudo 2008).



HUC-level wind erosion benefit categories are *dust cleaning* and *soil productivity*. Values are based on FPR-level[2] estimates.

[1]HUCs are watersheds defined by the U.S. Geological Survey's 8-digit hydrologic unit codes.
[2]FPRs are USDA's multi-state Farm Production Regions.

contingent valuation, hedonics, cost-based approaches, and factor-income approaches (Swinton et al. 2007). It is beyond this chapter to discuss these methods, but suffice it to say that each method is applicable to different situations and that the estimates derived have varying levels of confidence and comparability (Randall 2007).

In the only national-level estimate of conservation values with regional applications, Hansen and Ribaudo (2008) estimated the values of various soil conservation benefits. Although their focus was primarily on the impacts on end uses of water, they do provide a value for reducing soil erosion. Different values were derived for water and wind erosion, and these values varied by region (Figs. 1 and 2). Huszar (1989) estimated that off-site economic costs exceeded on-site costs from wind erosion and concluded that if public action were to be warranted, it should be aimed at reducing off-site impacts.

One of the issues in many valuation studies using willingness-to-pay measures is that while values are estimated, they do not address how values will change as supply and demand for that ecosystem good and service change. As an example, Loomis (2005) estimated values for outdoor recreation on public lands based on numerous studies. While these values may be valid, local conditions and the quantity of these goods and services nearby will affect these values.

In a study to compare the value of ecosystem services from restored versus native land, Dodds et al. (2008) estimated values based on a broad literature search. Estimated values for rangeland regions are shown in Table 6. It is important to note that they generally estimated lower values for restored lands. The implication of these values may be that society values the maintenance of native rangelands more than restored or that restored lands have not been shown to be as productive in producing these ecosystem goods and services as intact native rangelands.

Aesthetics are often cited as one of the important ecosystem services derived from rangelands. Most studies dealing with aesthetics have used contingent valuation. In one study that sought to actually quantify what ranch buyers would pay for a "quality-of-life" amenity that comes with owning the ranch, Torell et al. (2005b) found that ranch location, its scenic view, and the desirable lifestyle had more of an influence on ranch price than its potential to produce income. One of the implications of this is that while ranch owners may not respond to conservation practice

implementation to improve ranch income, they may respond to practices that enhance these factors.

Social Valuation

There is a small but growing literature on the valuation of ecosystem goods and services with metrics other than monetary valuation. These are presented here under social valuation since they seek to find alternative metrics for evaluating trade-offs among different products or outcomes.

One such approach has been to define a social-ecological system and solving a system of structural equations (Asah 2008). Finding the relationships between social systems and ecological systems remains a challenge. This method sought to relate a management goal with social knowledge of ecological responses. The results, however, are difficult to extrapolate because of the "place-specific nature of human-environment interactions" (Asah 2008).

Another system uses what are termed "holistic ecosystem health indicators" to integrate ecological, social, and interactive indicators (Munoz-Erickson et al. 2007). The ecological indicators use biophysical measurements; the social indicators use demographics, economics, and quality-of-life metrics; and the interactive indicators use land use practices, policy, and collaboration measurements. Each of the measurements is weighted to derive a measure of overall holistic ecosystem health.

RECOMMENDATIONS

Use of social and economic information can be incorporated into the NRCS conservation planning processes in a variety of ways. NRCS has done a commendable job of considering the ecosystem services in their planning and management processes. Acknowledgment of potential ecosystem services in the Conservation Practice Physical Effects Worksheets is an important step forward. We encourage the NRCS to continue to develop these worksheets, to refine their physical effects and rationale, and to define them on a more site-specific basis (i.e., by major land resource areas or ecological sites, as appropriate). We further recommend that the economic and cultural categories be expanded values of ecosystem services and social impacts at the individual, ranch or farm, and community levels.

TABLE 6. Estimated values of ecosystem services per native hectare per year (in 2005 dollars) (Dodds et al. 2008).

Ecosystem service	Great Plains		North American Deserts	
	Native	Restored	Native	Restored
Gas regulation	7	6	—	—
Disturbance regulation	7	7	2	1
Water supply	28	19	85	25
Nutrient cycling	22	15	60	18
Soil erosion control	241	175	237	65
Commodities	3 853	2 490	—	—
Biodiversity	46	50	—	—
Recreation	1 003	1 003	16	16

The method currently being used by NRCS to evaluate the benefits and costs contains the main features present in all the standard economic analyses (NPV) with the addition of values for selected ecosystem goods and services identified in each practice's description (H. Gordon, personal communication, 2008). The NRCS should continue to refine how it incorporates ecosystem goods and services into its conservation practice analysis. While the current economic analysis spreadsheet incorporates factors from the Physical Effects Worksheets and attempts to place monetary values on each item, justification for those monetary values needs to be developed and standardized. While the approach is sound, without a sound basis, values for the various ecosystem goods and services can easily be manipulated to justify any project. It should also be recognized that a complete benefit-to-cost assessment of selected conservation projects is not possible until valid estimates of economic value is assigned to presently unvalued ecosystem services.

The NRCS should seriously review its cost-share policies and requirements. If they are truly designed to pay for that portion of a conservation practice that benefits society, then the percentages should reflect that split. There are cost-share options that could be examined.

We recommend serious consideration of the first interpretation of cost share below:

1. Set cost share based on the split between expected private and public benefits. In some instances, this could range from 0% to 100%. Thus, rather than trying to estimate ecosystem goods and services values for each project, they could be determined for each practice through setting the appropriate cost share. Private landowners could then determine if their share of the total project cost would be covered by changes in their private benefits over the life of the project.
2. Determine whether conservation is a priority and use cost-share amounts to promote adoption of those practices with the highest public benefit. If the only purpose behind cost share is to get the private entity to become invested in the practice, the level of investment should be examined with a view toward the impacts that higher, the same, or lower cost shares would have on adoption rates.

KNOWLEDGE GAPS

Knowledge gaps in the social and economic realms of conservation practices are numerous. Here we cover a few specific research needs for economics, ecosystem services, and social science.

Economics of Conservation Practices

Since most rangeland conservation practices will be implemented by private ranchers throughout the western United States, it is necessary to understand how changes affect the overall economics of the ranching operation. Economic analysis of conservation practices can start with a basic efficiency estimate, such as present net worth, benefit-to-cost analysis, or internal rate of return. While that will provide a basic estimate of profitability, ranchers also need to understand how the change will affect their entire operation, and society needs to understand the larger-scale social benefits and costs.

Bighorn sheep near Gabbs, Nevada. (Photo: Tim Torell)

In terms of the actual conservation practice, the methodology for economic analysis as related to livestock production is well understood. Research that is needed at this level is knowledge about the physical responses (e.g., additional production and seasonality), the costs of inputs and outputs, and the timing of benefits and costs. Caton et al. (1960) noted that early attempts by agricultural economists participating in a West-wide regional research project to quantify the economics of rangeland management practices were hampered by a lack of response data. Potential benefits of many improvements could not be assessed because long-term studies had not been undertaken to quantify the forage and livestock responses that were realized from the various practices. This limitation continues.

The economic impacts of the conservation practice on the entire ranching operation require additional research. Most of the relationships are biological in nature (e.g., livestock production and forage production) and require knowledge about annual cycles and longer-term responses. Each livestock production cycle will have its own unique attributes, depending on geographic location, type of animal species, and production goals. In order to model within a year, these factors must be understood. When management changes with resulting changes in herd size, it may take several years for the ranch to come to a new equilibrium herd size. On the forage side, within-year variation of production affects the amount of feed available for the herd. Within the year and across years, temperature and precipitation will also affect the amount of feed available for livestock. Many of the conservation practices will affect the potential amount of forage production.

Social Aspects of Conservation Practices

There are many knowledge gaps related to social aspects of conservation practices.

The problem with many of these is that the knowledge that is needed is place based. What might be socially acceptable for a national program may not be at the local level. While there have been a few studies looking at adoption of conservation practices, these need to be done at more locations and specific to NRCS conservation practices on rangelands. In addition to location, such information for individuals, operations, and social systems will be useful in designing programs.

Ecosystem Goods and Services Valuation

Quantifying societal benefits and the economic value of previously nonquantified ecosystem services are the areas where economic evaluation of conservation practices is most lacking. As noted earlier, livestock production benefits do not justify the total cost of many conservation practices (McBryde et al. 1984; Lee et al. 2001; Torell et al. 2005a; Skaggs 2008). Economic evaluations are incomplete, and the assumption is made that ecosystem services not quantified in the analysis offset the excess cost of the practice not justified from marketable goods and services. This may not be the case. Only by quantifying and assigning economic value to selected ecosystem services that are currently only qualitatively noted can a complete economic assessment be made. Quantifying benefits accruing to the public at large could also be used to justify cost-share percentages on a case-by-case (or location-by-location) basis. This would mean an expanded use of nonmarket valuation techniques (see Champ et al. 2003). We note that while it would be advantageous to know how people value each of the goods and services produced by different conservation practices, it is probably neither likely nor feasible. Yet expanding the economic analysis to quantify all potential benefits has several important implications. First, the quantification may show a very positive benefit-to-cost ratio from society's point of view, suggesting that even more should be done. In other cases, it may demonstrate that the conservation practice is not justified. If not valued by society, based on economics, transfer payments and willingness to pay, inaction, and a deteriorated ecosystem may be the preferred state. Dismal federal and state budget situations highlight that trade-offs exist.

Most of the knowledge base on the values of nonmarket goods and services is time and space specific given the methodologies currently in use. Aggregation of numerous studies can provide value ranges that various researchers have estimated using methods such as travel cost, contingent valuation, or hedonic models. Whether these are appropriate for the NRCS to use in evaluating their conservation practices through extrapolation is a question for investigation. The basic knowledge gaps are the values for each ecosystem good or service in each location and at the specific time.

CONCLUSIONS

Use of Economic Information in Resource Planning and Management

Economic values for market and nonmarket ecosystem goods and services will vary by location and time. What might be available at the scale necessary for ranch-level planning are indicators of relative values. There have been many studies of individual benefits for specific goods and services. Those that exist in markets provide what people are truly willing and able to pay. Some methods, such as hedonic pricing and willingness-to-pay studies, can help determine values of specific characteristics.

Relative values of the various goods and services can provide information to planners and managers if they are collected appropriately and in a consistent manner. There are numerous peer-reviewed articles for many of the conservation practices that have done an economic analysis of the specific practice comparing the cost of the practice with the estimated market benefits. We have not reviewed those studies because they are not particularly enlightening for this project. The method currently being used by the NRCS to evaluate the benefits and costs contains the main features present in all the standard analyses with the addition of values for the various ecosystem goods and services identified in each practice's description (H. Gordon, personal communication, 2008).

If economic feasibility of various conservation practices is important, it would be prudent for NRCS to become seriously involved in finding ways to assess the economic value of ecosystem goods and services beyond those found in

"Quantifying societal benefits and the economic value of previously nonquantified ecosystem services are the areas where economic evaluation of conservation practices is most lacking."

Old cabin and scenic view, Owyhee County, Idaho. (Photo: John Tanaka)

the marketplace. As noted, when considering only forage value or livestock production from rangelands as the primary benefit, many conservation practices will not show a favorable benefit-to-cost ratio for conservation programs. As the ecosystem goods and services are considered and valued, the NRCS can allocate the conservation practice costs to the private landowner and to taxpayers through its cost-share mechanism based on the expected proportion of benefits going to the different parties. For example, if a prescribed grazing practice does not increase livestock production and net returns but produces significant social benefits, the taxpayers may be allocated a larger proportion of the costs of implementation than if livestock benefits were higher. Some of the social benefits may go to the private landowner, and this should also be taken into account (e.g., maintaining a way of life). At present, while cost-share mechanisms would seem to implicitly recognize these social benefits, there is little information to justify the cost-share percentages on a case-by-case (or location-by-location) basis. It is not reasonable to assume that the social benefits from a given conservation practice is the same everywhere or that they offset substantial practice costs in many cases.

Quantitative estimates of the value of ecosystem service are largely nonexistent, but quantification of traditional market values are lacking as well. Efforts to quantify the economics of rangeland management practices have been hampered by a lack of response data. Long-term range and grazing studies that monitored the production response of management practices were found to be a major shortcoming of economic assessments of rangeland management practices that have continued since the 1960s. These economic assessments of rangeland management practices are based on very limited data and usually use simulated biophysical data. Long-term range and grazing studies are becoming even less common. Response relationships among conservation practices and other ecosystem goods and services are even rarer or nonexistent.

Use of Social Information in Resource Planning and Management

Social values and attitudes have clearly impacted adoption of rangeland management practices and how rangeland policy and management has progressed. As an example, Young and Clements (2009, p. 178.)

concluded, "Public rangeland management agencies did not drop the use of herbicides because they were afraid of the environmental consequences of using pesticides; they dropped them because they were afraid of comments from a highly vocal but not necessarily knowledgeable portion of the general public. Congress stopped appropriating money for the improvement of publicly owned rangelands to avoid criticism from environmental groups." The numerous other examples described above clearly show that social factors have motivated land managers to behave in ways beyond profit-maximizing behavior.

If social information is going to be used in resource planning and management, social indicators need to be added to the list of benefits along with a description of how to use and interpret the indicators. Our assumption is that most resource managers do not know what indicators would be appropriate or how to use them in decision making. The significant lack of social research on the effects of the conservation practices on landowners will make implementation of this recommendation difficult in any sort of quantitative manner.

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