

Natural Resources and Environmental Issues

Volume 16

Issue 1 *Shrublands: Wildlands and Wildlife Habitats*

Article 1

January 2011

15th Wildland Shrub Symposium - Shrublands: Wildlands and Wildlife Habitats, 2008 June 17-19, Bozeman, MT

Carl L. Wambolt

Stanley G. Kitchen

Michael R. Frisina

Bok Sowell

Richard B. Keigley

See next page for additional authors

Wambolt, Carl L.; Kitchen, Stanley G.; Frisina, Michael R.; Sowell, Bok; Keigley, Richard B.; Palacios, Patsy; and Robinson, Jill (2011) "15th Wildland Shrub Symposium - Shrublands: Wildlands and Wildlife Habitats, 2008 June 17-19, Bozeman, MT," *Natural Resources and Environmental Issues*: Vol. 16, Article 1.
Available at: <http://digitalcommons.usu.edu/nrei/vol16/iss1/1>

This Article is brought to you for free and open access by the Quinney Natural Resources Research Library, S.J. and Jessie E. at DigitalCommons@USU. It has been accepted for inclusion in Natural Resources and Environmental Issues by an authorized administrator of DigitalCommons@USU. For more information, please contact digitalcommons@usu.edu.

Take a 1 Minute Survey- <http://www.surveymonkey.com/s/BTVT6FR>



Carl L. Wambolt, Stanley G. Kitchen, Michael R. Frisina, Bok Sowell, Richard B. Keigley, Patsy Palacios, and
Jill Robinson

Proceedings



SHRUBLANDS:

wildlands and wildlife habitats

15th Wildland Shrub Symposium
2008 June 17-19; Bozeman, MT



Natural Resources and Environmental Issues

**Volume XVI
2011**

Suggested Citation:

Wambolt, Carl, L.; Kitchen, Stanley G.; Frisina, Michael R.; Sowell, Bok F.; Keigley, Richard B.; Palacios, Patsy K.; Robinson, Jill A., comps. 2011. Proceedings – Shrublands: Wildlands and Wildlife Habitats; 2008 June 17-19; Bozeman, MT. Natural Resources and Environmental Issues, Volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan Utah, USA.

Abstract

The 35 papers in this proceedings are divided into four sections; the first includes an introduction to the symposium theme of *Shrublands as wildlands and wildlife habitat*, along with keynote addresses discussing geographic affiliations of eastern Montana's "Great Plains Flora" and methodology for surveying mule deer winter range habitat use and condition. The next two sections cluster papers on wildlife habitat and ecological relationships. These sections provide a diverse sampling of topics examining the nature and impacts of intra- and inter-trophic relationships among plants and associated species of western North American shrublands. Papers in the final section present assessments of various sampling and monitoring methodologies applicable to shrublands.

Keywords: wildland shrubs, sagebrush, herbivory, browsing, fire, monitoring

Cover photos:

Sage-grouse taken by Jennifer Woodward,
antelope, elk, and deer taken by Carl Wambolt



15th Wildland Shrub Symposium
Shrublands: Wildlands and Wildlife Habitats
2008 June 17-19
Bozeman, MT

Compilers:
Carl L. Wambolt
Stanley G. Kitchen
Michael R. Frisina
Bok Sowell
Richard B. Keigley
Patsy Palacios
Jill Robinson

NATURAL RESOURCES AND ENVIRONMENTAL ISSUES

VOLUME XVI
2011



**Montana Fish,
Wildlife & Parks**



Series Forward

Natural Resources and Environmental Issues (NREI) is a series devoted primarily to research in natural resources and related fields. This series is published by the S.J. and Jessie E. Quinney Natural Resources Research Library, College of Natural Resources, Utah State University, Logan, Utah 84322-5260.

The Quinney Library recognizes its role in educating natural resources leaders who can then provide the guidance needed to increase the production of the earth's renewable resources while sustaining / enhancing the global environment. This publication, first published in 1993, is a monographic series that addresses relevant current topics. The journal is published in a series of volumes, each addressing a single topic and often associated with an organized symposium. Publication in NREI is by invitation only.

Volumes of this series will be sent upon request. Single issues are \$25.00 in the United States and \$30.00 elsewhere. Payment can be made by credit card, check, money order, or purchase order, payable in U.S. dollars to the Quinney Library. Order information is located in the back of this publication.

Articles and information appearing in Natural Resources and Environmental Issues are the property of the specified authors. They may be reprinted with written permission provided that no endorsement of a specific commercial product or firm is stated or implied. Authors, the College of Natural Resources, the Quinney Library, and Natural Resources and Environmental Issues must be credited.

This is an open access journal which means that all content is freely available without charge to the user or his/her institution. Users are allowed to read, download, copy, distribute, print, search, or link to the full texts of the articles in this journal without asking prior permission from the publisher or the author. This is in accordance with the BOAI definition of open access. Visit us at <http://digitalcommons.usu.edu/nrei/>

Contact Information:

Address: NREI, 5260 Old Main Hill,
Logan, UT 84322
Phone: 435.797.2464
Fax: 435.797.3798
E-mail:
quinneylibrary@aggiemail.usu.edu

©2011 by the Quinney Library
College of Natural Resources
Utah State University

ISSN 1069-5370

Shrub Research Consortium

Federal Government:

USDA Forest Service, Rocky Mountain Research Station, Grassland, Shrubland, Desert Ecosystems Program:

Shrub Sciences Laboratory*, Provo, UT, Stanley G. Kitchen (chair)

Forestry Sciences Laboratory, Reno, NV, Robin J. Tausch

Forestry Sciences Laboratory, Albuquerque, NM, Rosemary L. Pendleton

USDA Agricultural Research Service:

Jornada Experimental Range, Las Cruces, NM, Mary Lucero

Forage and Range Research Laboratory, Logan, UT Thomas Monaco

High Plains Grassland Research Station, Cheyenne, WY, D. Terrance Booth

Exotic and Invasive Weed Research Unit*, Reno, NV

USGS Northern Rocky Mountain Science Center, Bozeman, MT, Richard B. Keigley

State Governments:

Utah Department of Natural Resources, Division of Wildlife Resources*, Great Basin Research

Center, Ephraim, UT, Danny Summers

Montana Fish, Wildlife and Parks, Butte MT, Michael R. Frisina

Universities:

Brigham Young University*, Provo, UT, Steve Petersen

Utah State University*, Logan, UT, Eugene W. Schupp

University of California, Los Angeles, CA, Philip W. Rundel

Colorado State University, Fort Collins, CO

University of Idaho, Moscow, ID

University of Montana, Missoula, MT, Donald J. Bedunah

Montana State University, Bozeman, MT, Bok Sowell

University of Nevada, Reno, NV, Barry L. Perryman

University of Nevada, Las Vegas, NV

Oregon State University, Corvallis, OR

New Mexico State University, Las Cruces, NM, Anders F. Cibils

Texas A & M System, Texas Agricultural Experiment Station, San Angelo, TX, R. J. (Jim) Ansley

Texas Tech University, Lubbock, TX, David B. Wester

University of Utah, Salt Lake City, UT, John S. Sperry

Weber State University, Ogden, Utah, UT, Barbara A. Wachocki

University of Wyoming, Laramie WY, Ann L. Hild

Private:

National Security Technologies, Las Vegas, NV, Kent Ostler

*Charter members

The Shrub Research Consortium (SRC) was formed in 1983 with five charter members (see list). Over time, SRC has grown to its present size of 27 institutional members. The SRC charter had three principal objectives: (1) develop plant materials for shrubland rehabilitation; (2) develop methods of establishing, renewing, and managing shrublands in natural settings; and (3) assist with publication and dissemination of research results. These objectives have been met by a series of symposia sponsored by the Consortium and partners. This publication is the 15th in the series. The 14 previous symposia are listed on the next page. Proceedings of all publications to date have been published by the U.S. Department of Agriculture, Forest Service, Intermountain Research Station and Rocky Mountain Research Station. Each symposium has had a theme, but the executive committee has encouraged attendance and participation by shrubland ecosystem biologists and managers with wider interests than any particular symposium theme—the heart of the Consortium's programs are wildland shrub ecosystem biology, research, and management.

Availability of Previous Wildland Shrub Symposia Proceedings

First: Tiedemann, A. R.; Johnson, K. L., compilers. 1983. Proceedings—research and management of bitterbrush and cliffrose in western North America; 1982 April 13-15; Salt Lake City, UT. General Technical Report INT-152. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station, 279 p. Out of print—available from National Technical Information Service as document PB83261537. Contact NTIS at: (800) 553-6847, or order online at: www.ntis.gov.

Second: Tiedemann, A. R.; McArthur, E. D.; Stutz, H. C.; Stevens, R.; Johnson, K. L., compilers. 1984. Proceedings—symposium on the biology of Atriplex and related chenopods; 1983 May 2-6; Provo, UT. General Technical Report INT-172. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 309 p. Out of print—available from National Technical Information Service as document PB85116358. Contact NTIS at: (800) 553-6847, or order online at: www.ntis.gov.

Third: McArthur, E. D.; Welch, B. L., compilers. 1986. Proceedings—symposium on the biology and management of *Artemisia* and *Chrysothamnus*; 1984 July 9-13; Provo, UT. General Technical Report INT-200. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 398 p. Out of print—available from National Technical Information Service as document PB86182318. Contact NTIS at: (800) 553-6847, or order online at: www.ntis.gov.

Fourth: Provenza, F. D.; Flinders, J. T.; McArthur, E. D., compilers. 1987. Proceedings—symposium on plant herbivore interactions; 1985 August 7-9; Snowbird, UT. General Technical Report INT-222. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 179 p. A few copies are available from the Rocky Mountain Research Station; otherwise available from National Technical Information Service as document PB90228578. Contact NTIS at: (800) 553-6847, or order online at: www.ntis.gov.

Fifth: Wallace, A.; McArthur, E. D.; Haferkamp, M. R., compilers. 1989. Proceedings—symposium on shrub ecophysiology and biotechnology; 1987 June 30-July 2; Logan, UT. General Technical Report INT-256. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 183 p. Available from Rocky Mountain Research Station: <http://www.fs.fed.us/rm/publications/>.

Sixth: McArthur, E. D.; Romney, E. M.; Smith, S. D.; Tueller, P. T., compilers. 1990. Proceedings—symposium on cheatgrass invasion, shrub die-off, and other aspects of shrub biology and management; 1989 April 5-7; Las Vegas, NV. General Technical Report INT-276. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 351 p. Out of print—available from National Technical Information Service as document PB91117275. Contact NTIS at: (800) 553-6847, or order online at: www.ntis.gov.

Seventh: Clary, W. P.; McArthur, E. D.; Bedunah, D.; Wambolt, C. L., compilers. 1992. Proceedings—symposium on ecology and management of riparian shrub communities; 1991 May 29-31; Sun Valley, ID. General Technical Report INT-289. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 232 p. Out of print—available from National Technical Information Service as document PB92227784. Contact NTIS at: (800) 553-6847, or order online at: www.ntis.gov.

Eighth: Roundy, B. A.; McArthur, E. D.; Haley, J. S.; Mann, D. K., compilers. 1995. Proceedings: wildland shrub and arid land restoration symposium; 1993 October 19-21; Las Vegas, NV. General Technical Report INT-GTR-315. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 384 p. Available from the Rocky Mountain Research Station: <http://www.fs.fed.us/rm/publications/> or you can download the publication at: http://www.fs.fed.us/rm/pubs_int/int_gtr315.html.

Ninth: Barrow, J. R.; McArthur, E. D.; Sosebee, R. E.; Tausch, R. J., compilers. 1996. Proceedings: shrubland ecosystem dynamics in a changing environment; 1995 May 23-25; Las Cruces, NM. General Technical Report INT-GTR-338. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 275 p. Available from the Rocky Mountain Research Station: <http://www.fs.fed.us/rm/publications/>.

Tenth: McArthur, E. D.; Ostler, W. K.; Wambolt, C. L., compilers. 1999. Proceedings: shrubland ecosystem ecotones; 1998 August 12-14; Ephraim, UT. Proceedings RMRS-P-11. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 299 p. Available from the Rocky Mountain Research Station: <http://www.fs.fed.us/rm/publications/>.

Eleventh: McArthur, E. D.; Fairbanks, D. J., compilers. 2001. Shrubland ecosystem genetics and biodiversity: proceedings; 2000 June 13-15; Provo, UT. Proceedings RMRS-P-21. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 365 p. Available from the Rocky Mountain Research Station: <http://www.fs.fed.us/rm/publications/> or you can download the publication at: http://www.fs.fed.us/rm/pubs/rmrs_p021.html.

Twelfth: Hild, A. L.; Shaw, N. L.; Meyer, S. E.; Booth, D. T.; McArthur, E. D., compilers. 2004. Seed and soil dynamics in shrubland ecosystems: proceedings; 2002 August 12-16; Laramie, WY. Proceedings RMRS-P-31. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 216 p. Available from the Rocky Mountain Research Station: <http://www.fs.fed.us/rm/publications/> or you can download the publication at: http://www.fs.fed.us/rm/pubs/rmrs_p031.html.

Thirteenth: Sosebee, R. E.; Wester, D. B.; Britton, C. M.; McArthur, E. D.; Kitchen, S.G., comps. 2007. Proceedings: shrubland dynamics--fire and water; 2004 August 10-12; Lubbock, TX. Proceedings RMRS-P-47. Fort Collin, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 173 p. Available from the Rocky Mountain Research Station: http://www.fs.fed.us/rm/pubs/rmrs_p047.html.

Fourteenth: Kitchen, S. G.; Pendleton, R. L.; Monaco, T. A.; Vernon, J., comps. 2008. Proceedings-Shrublands under fire: disturbance and recovery in a changing world; 2006 June 6-8; Cedar City, UT. Proc. RMRS-P-52. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 190 p. Available from the Rocky Mountain Research Station: http://www.fs.fed.us/rm/pubs/rmrs_p052.html.

Notice of Retraction:

Based upon information uncovered in an investigation conducted by the University of Nevada, Las Vegas, The Shrub Research Consortium (SRC) Executive Committee finds it necessary to retract the article, “Soil responses to human recreational activities in a blackbrush (*Coleogyne ramosissima* Torr.) shrubland” published in, Proceedings: Shrubland dynamics – Fire and Water, RMRS-P-47, 2007. The University has indicated that the author, Simon A. Lei, admitted to research misconduct in the form of fabricating the data used in this article.

In addition, the University suggested that there was reason to believe that the author may have fabricated data for these additional articles and requested that they also be retracted.

“Ecological leaf anatomy of seven xerophytic shrub species in southern Nevada”, Proceedings: Shrubland Ecotones, RMRS-P-11, 1999.

“Ecology of psammophytic plants in the Mojave, Sonoran, and Great Basin Deserts”, Proceedings: Shrubland Ecotones, RMRS-P-11, 1999.

“Intraspecific variation in leaf anatomy of blackbrush (*Coleogyne ramosissima* Torr.)”, Proceedings: Shrubland dynamics – fire and water, RMRS-P-47, 2007.

“Radial symmetry in the white bursage (*Ambrosia dumosa*) in the Mojave Desert”, Proceedings: Shrublands under fire: disturbance and recovery in a changing world, RMRS-P-52, 2008.

The SRC Executive Committee is committed to publishing papers representing diverse viewpoints as long as the research studies that are cited in support of those papers are conducted with the highest ethical standards. It is troubling that anyone might attempt to use any of the symposia sponsored by the SRC, or the associated published proceedings, to achieve unethical purposes.

Consequently, with this notice the SRC Executive Committee announces the retraction of each of the five articles previously published in SRC sponsored symposium proceedings listed above. In addition, these articles have been removed from Treesearch, an on-line Forest Service publication library service.

Publisher’s note: Papers in this and other proceedings sponsored by the Shrub Research Consortium were reviewed by the compilers for format and style. Authors are responsible for content.

Table of Contents

Introduction and Overview

Shrublands: Wildlands and Wildlife Habitats	1
<i>Carl L. Wambolt</i>	
Great Plains Flora? Plant Geography of Eastern Montana's Lower Elevation Shrub-grass Dominated Vegetation	3
<i>Matt Lavin and Catherine Seibert</i>	
Long-term Browsing Impacts on Montana Ungulate Ranges	15
<i>Scott K. Thompson and Carl L. Wambolt</i>	

Wildlife Habitat in Shrublands

Sage-grouse (<i>Centrocercus urophasianus</i>) Habitat in Central Montana	21
<i>Jennifer Woodward, Carl Wambolt, Jay Newell, and Bok Sowell</i>	
Potential and Pitfalls of Prescribed Burning Big Sagebrush Habitat to Enhance Nesting and Early Brood-Rearing Habitats for Greater Sage-Grouse	27
<i>Jeffrey L. Beck, J. Garrett Klein, Justin Wright, and Kenneth P. Wolfley</i>	
Response of Obligate Birds to Mechanical Manipulations in a Sagebrush Ecosystem Near Gunnison, Colorado	33
<i>Patrick A. Magee, Jason Brooks, Nick Hirsch, and Tyler L. Hicks</i>	
Effect of Browsing Following Wildfire in the Missouri Breaks	45
<i>Richard B. Keigley and Kent Undlin</i>	
Quaking Aspen Ecology on Forest Service Lands North of Yellowstone National Park	51
<i>David S. Kimble, Daniel B. Tyers, and Bok F. Sowell</i>	
Restoring Aspen Riparian Stands with Beaver on the Northern Yellowstone Winter Range	59
<i>Samuel D. McColley, Dan B. Tyers, and Bok F. Sowell</i>	
Sagebrush-Obligate Passerine Response to Ecological Site Characteristics	63
<i>Mary I. Williams, Thomas L. Thurow, Ginger B. Paige, Ann L. Hild, and Kenneth G. Gerow</i>	
Elk and Fire Impacts on Mountain Big Sagebrush Range in Yellowstone	73
<i>Carl L. Wambolt and Reyer J. Rens</i>	

Ecological Relationships in Shrublands

Post-fire Recovery of Wyoming Big Sagebrush Steppe in Central and Southeast Montana	79
<i>Stephen V. Cooper, Peter Lesica, and Greg M. Kudray</i>	
Dynamics of Western Juniper Woodland Expansion into Sagebrush Communities in Central Oregon ...	89
<i>Mary M. Rowland, Lowell H. Suring, Robin J. Tausch, Susan Geer, and Michael J. Wisdom</i>	

Relationship of Wyoming Big Sagebrush Cover to Herbaceous Vegetation	101
<i>Bok F. Sowell, Carl L. Wambolt, Jennifer K. Woodward, and Vanessa R. Lane</i>	
Vegetation Dynamics at a Mojave Desert Restoration Site, 1992 to 2007	105
<i>Jeffrey E. Ott, E. Durant McArthur, and Stewart C. Sanderson</i>	
Converting Crested Wheatgrass Stands to Enhance Big Sagebrush: A Literature Review	121
<i>Krystle A. Pehrson and Bok F. Sowell</i>	
Clarifying Potential Successional Trajectories in Sagebrush Communities Historically Seeded with Crested Wheatgrass	131
<i>Kevin L. Gunnell, Justin R. Williams, and Thomas A. Monaco</i>	
Effect of Atmospheric CO ₂ Levels on Nutrients in Cheatgrass Tissue	139
<i>Robert R. Blank, Tye Morgan, Lewis H. Ziska, and Robert H. White</i>	
Influence of Plant Invasion on Seed Chemistry of Winterfat, Green Rabbitbrush, Freckled Milkvetch, Indian Ricegrass, and Cheatgrass	145
<i>Robert R. Blank, Tye Morgan, and Fay Allen</i>	
Effects of Spring Prescribed Fire in Expanding Pinyon-Juniper Woodlands on Seedling Establishment of Sagebrush Species	149
<i>David I. Board, Jeanne C. Chambers, and Joan G. Wright</i>	
Examining Endophyte Interactions within Fourwing Saltbush (<i>Atriplex canescens</i>)	159
<i>Mary E. Lucero, Jerry R. Barrow, Isaac Reyes-Vera, and Ruth Sedillo</i>	
Seeded Native Shrub Establishment on Disturbed Sites in Southwestern Wyoming	165
<i>James S. Jacobs, Susan R. Winslow, Karen J. Clause, and Roger Hybner</i>	
Incorporating Historical Ecosystem Diversity into Conservation Planning Efforts in Grass and Shrub Ecosystems	171
<i>Amy C. Ganguli, Jonathan B. Haufler, Carolyn A. Mehl, and Jimmie D. Chew</i>	
Short-Term Response of Shrubs, Graminoids, and Forbs to Mechanical Treatment in a Sagebrush Ecosystem in Colorado	179
<i>Adam C. Payton, Sandra J. Hayes, Sandra M. Borthwick, and Russell D. Japuntich</i>	
Displacement of Native Riparian Shrubs by Woody Exotics: Effects on Arthropod and Pollinator Community Composition	185
<i>Rosemary L. Pendleton, Burton K. Pendleton, and Deborah Finch</i>	
Post-burn Resprouting in Bigtooth Maple (<i>Acer grandidentatum</i>): A Case Study	197
<i>Beth L. Corbin and Douglas H. Page¹</i>	

¹This paper was presented as a poster at the Fourteenth Wildland Shrub Symposium, Shrublands Under Fire: Disturbance and Recovery in a Changing World; June 6-8, 2006 in Cedar City, UT. Because it was inadvertently left out of the corresponding published proceedings it is included here with the authors' permission.

Sampling and Monitoring in Shrublands

Big Sagebrush in Pinyon-juniper Woodlands: Using Forest Inventory and Analysis Data as a Management Tool for Quantifying and Monitoring Mule Deer Habitat	201
<i>Chris Witt and Paul L. Patterson</i>	
A Landscape Level Habitat Survey of Mule Deer Winter Range in Eastern Montana	209
<i>William H. Thompson, Paul L. Hansen, and Michael R. Frisina</i>	
A Process to Monitor and Manage Ungulate Browsing Pressure	215
<i>Richard B. Keigley and Michael R. Frisina</i>	
Getting the Biggest Bang for Your Buck: Wildlife Monitoring on Shrublands of the Nevada Test Site	223
<i>Derek B. Hall and Paul D. Greger</i>	
Long-Term Changes in Southern Utah Upland Shrub Communities: A Study in Repeat Photography ...	229
<i>Charles E. Kay and Chad R. Reid</i>	
Classification and Management of Upland, Riparian, and Wetland Sites of USDI Bureau of Land Management's Miles City Field Office, Eastern Montana USA	239
<i>Paul L. Hansen, William H. Thompson, Ray Smith, and Todd Yeager</i>	
Ecological Assessment of Sagebrush Grasslands in Eastern Wyoming	249
<i>Amy C. Ganguli, Jonathan B. Haufler, Carolyn A. Mehl, and Scott D. Yeats</i>	
Comparing Vegetation Monitoring Methods in Shrublands: How Valuable is Grant's Method in Shrub Communities?	259
<i>Matt M. Church, Mary I. Williams, Ann L. Hild, and Ginger B. Paige</i>	
A Metric System for Evaluating Off-site Mitigation for Ecosystem Services and Wildlife Habitat in Sagebrush Ecosystems	265
<i>Jonathan B. Haufler and Lowell H. Suring</i>	

SHRUBLANDS: Wildlands and Wildlife Habitats

Carl L. Wambolt

The fifteenth symposium sponsored by the Shrub Research Consortium on the biology and management of shrubs was held June 17-19, 2008 in Bozeman, Montana. This symposium furthers the objective of the Shrub Research Consortium to facilitate the spread of knowledge relating to shrub biology and management. The biological relationships of shrubs and shrublands including their value as wildlands and wildlife habitats was the primary theme of the 50 papers and posters presented during the symposium. While presentations were not restricted to these topics, the majority were concerned with the aspects of shrubs and shrublands that are important to understanding relatively unaltered (wild) shrub communities and their attributes that make them important, if not critical, as wildlife habitats. Many of the presentations focused on the Greater Yellowstone ecosystem and surrounding region, but they included many other diverse locations in western North America.

The mid-symposium field tours were also arranged to emphasize research and management topics on important wildland wildlife habitats. One tour visited the Northern Yellowstone Winter Range (NYWR). The NYWR is world famous for its large ungulate and predator populations and their interrelationships. For at least a century the interrelationships of wildlife and shrub habitats have been a focal point of interest and keenly followed by ecologists on the NYWR. The complicated ecological relationships on the NYWR have served as classroom material for decades throughout the world. The NYWR is approximately 60 percent in Yellowstone National Park with the remainder in Montana. Participants were provided a perspective of factors influencing NYWR shrub habitats since settlement of Montana's Yellowstone River Valley and creation of Yellowstone National Park in 1872. The alternate tour traveled to the Mount Haggin Wildlife Management Area (WMA) south of Anaconda, Montana. The Mount Haggin WMA is state-owned and managed by Montana Fish, Wildlife & Parks. The 58,000 acre property is habitat for most of Montana's big-game species, with emphasis on elk, moose, and mule deer. The Mount Haggin WMA is surrounded by similar public land habitats. Participants learned of the region's history and the forces that shaped the wildlife habitats. Research documenting the effect of browsing history on the Mount Haggin WMA was discussed. Participants were offered an opportunity to learn details about the methods used to study browsing history on tall shrubs.

These proceedings include manuscripts from 33 of the papers and posters presented at the symposium. They are divided into four broad sections by subject that includes: 1) Introduction and Overview, 2) Wildlife Habitat in Shrublands, 3) Ecological Relationships in Shrublands, and 4) Sampling and Monitoring in Shrublands. One additional paper presented at the Fourteenth Wildland Shrub Symposium in Cedar City, UT (2006) was inadvertently omitted from the corresponding published proceedings and is included here in the Ecological Relationships in Shrublands section.

As with any undertaking like this symposium, many people and organizations are to be thanked. I thank co-compilers Mike Frisina, Bok Sowell, and Richard Keigley for their help with planning, conductance, and preparation of the proceedings. Sandra Voegeli was invaluable in editing manuscripts to insure all manuscripts were ready for publication. Session conveners Bok Sowell, Carolyn Johnson-Nistler, and Mike Frisina were very effective. While I led the field to the Northern Yellowstone Winter Range, Richard Keigley led the very stimulating field trip to the Mount Haggin WMA. Sponsors of the symposium include the US Forest Service, Rocky Mountain Research Station; Montana State University (Dept. of Animal and Range Sciences); Montana Fish, Wildlife & Parks Department and the Montana Chapter of the Foundation for North American Wild Sheep.

Great Plains Flora? Plant Geography of Eastern Montana's Lower Elevation Shrub-grass Dominated Vegetation

Matt Lavin¹ and Catherine Seibert¹

ABSTRACT

Montana plant geography at elevations below montane forests broadly includes open shrub-grass or ponderosa pine dominated dry sites and riparian-wetland systems. In contrast to conventional wisdom, the floristic composition of these settings in eastern Montana does not reflect a strong Great Plains influence. State and county geographical distribution patterns suggestive of an influence of the Great Plains flora on that of eastern Montana involve only 52 species of mostly uncommon and narrowly distributed species of dicot forbs that do not compose a common type of characteristic Great Plains plant community in the state. In addition, the floristic similarity of the grass family, *Poaceae*, which is very diverse in open dry vegetation in Montana, reveals that Montana shares many more grass species in common with Utah than with the adjacent Great Plains state of South Dakota. Instead of the Great Plains biome, the low elevation flora and vegetation of Montana appears to be part of or influenced by the Pacific Northwest, Boreal, and the Intermountain biome. From the perspective of plant identification, the volumes of the Intermountain Flora works as well as or better across the state of Montana at low elevation dry settings compared to the Flora of the Great Plains or the Flora of the Pacific Northwest. The flora and vegetation of the open dry settings of eastern Montana is generally characteristic of the Intermountain sagebrush steppe and this should be considered in "prairie" restoration programs, especially when large ungulates like bison are proposed for reintroduction.

INTRODUCTION

The Vascular Plants of Montana (Dorn 1984), the Flora of the Great Plains (McGregor and Barkley 1986), the Pacific Northwest Flora (Hitchcock and Cronquist 1973), and the Intermountain Flora (Cronquist and others 1972) all show maps that delineate a floristic or vegetation division in the state of Montana that separates the mountains and foothills generally in the western one-third from the lower elevation "plains" in the eastern two-thirds of the state. The Flora of the Great Plains encompasses all plant species from the eastern two thirds of Montana, whereas the Flora of the Pacific Northwest includes all Montana plant species. The Intermountain Flora was developed without a consideration of the Montana flora. In alignment with the Potential Natural Vegetation of the Conterminous United

States (Küchler 1964), the Grassland and Shrubland Habitat Types of Western Montana (Mueggler and Steward 1980) includes a similar map and explicitly excludes eastern Montana from the classification system of shrub-grass dominated vegetation in western Montana because of its putative fundamental differences in physiognomy and floristic composition.

With this knowledge, the first author, inexperienced with field work in eastern Montana, offered to assess the diversity and quality of the flora and vegetation of lands in central Phillips County, which were being purchased and managed by the American Prairie Foundation (APF) in conjunction with the World Wildlife Fund to reintroduce bison and other mammals to the northern Great Plains (<http://www.americanprairie.org/>). The APF plans to restore some of the original conditions of the Great Plains ecosystem in eastern Montana. They clearly invoke a conceptual model that includes eastern Montana harboring Great Plains flora and vegetation.

Instead of finding grasslands dominated by an abundance of warm season grass species during that initial vegetation reconnaissance of APF lands during 2003, what predominated the native vegetation in dry open settings in much of Phillips and adjacent Counties were a large diversity of cool season Palouse prairie grasses (for example *Agropyron spicatum*, *Festuca idahoensis*, *Koeleria macrantha*, etc.), *Artemisia tridentata*, and many other shrubs and forbs well known from the Intermountain west. For plant identification, the Flora of the Great Plains (McGregor and Barkley 1984) was difficult to use because it contained many families, genera, and species that were either lacking or not diverse in Montana. In contrast, the volumes of the Intermountain flora worked very well, especially for the identification of the grass species. This preliminary floristic evidence suggested that much of eastern Montana was more of the Intermountain than Great Plains biome.

Further reconnaissance of the flora and vegetation of eastern Montana revealed extensive forests and woodland dominated by *Pinus ponderosa* var. *scopulorum*, sagebrush steppe harboring a large diversity of short statured cool rather than tall warm season grasses, riparian areas with species characteristic of eastern deciduous forests (for example *Fraxinus pennsylvanica*, *Quercus macrocarpa*, and *Ulmus americana*), or vegetation with widespread constituents (for example *Acer negundo* and *Populus fremontii*). Species characteristic of the Great Plains, such

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Matt Lavin and Catherine Seibert, MSU Herbarium (MONT), Department of Plant Sciences and Plant Pathology, Montana State University, Bozeman, MT 59717, Email: mlavin@montana.edu, ubics@montana.edu

tall statured warm season grasses or the cool season porcupinegrass, were difficult to locate at all or in high abundance. This initial finding prompted a more detailed investigation into the geography of Montana's vascular plants in order to determine whether its topography with mountains in the west and plains in the east actually imposes an ecological and geographic distinction on the flora of the state.

METHODS

Four types of data were gathered to test the hypothesis that the eastern low elevation flora of Montana is part of or significantly influenced by the Great Plains biome. First, geographical distributions of Great Plains species occurring in eastern Montana were inspected. Second, eastern Montana vegetation types dominated by Great Plains species or associations were identified. Third, the grasses of Montana were inspected for relative similarity to Great Plains versus Intermountain grasses. Finally, the North American distributions of species dominating low elevation habitats in Montana were inspected for geographical patterns in order to identify the relative influence of the Great Plains compared to other North American biomes such as that of the Intermountain or Pacific Northwest.

Great Plains Species Occurring in Eastern Montana

A list of plant species potentially characteristic of eastern Montana was derived primarily from Dorn (1984) by including all species assigned only to the geographic categories of "central" or "east" but not "west". This initial list was scrutinized for those species having a county distribution restricted to the eastern two-thirds of Montana. County distributions were derived from plant specimens deposited at the Montana State University Herbarium (MONT) and the University of Montana Herbarium (MONTU). This distilled list was further refined by retaining species with a distribution in North America that was primarily from the central corridor of states and provinces in Canada and the United States. Distributions among states and provinces in the USA and Canada were taken from the PLANTS Database (USDA, NRCS, 2008). This final list was then considered the extent of the influence of the Great Plains biome on eastern Montana vegetation.

Eastern Montana Vegetation Types Dominated by Great Plains Species

Plant species that characterize large scale vegetation or habitat types in Montana (Pfister and others 1977; Mueggler and Steward 1980; Montana Natural Heritage Program 2002) were especially scrutinized for their distribution both within Montana and throughout North America. Literature bearing specifically on the Rocky

Mountain flora and vegetation was not considered during this analysis because of the focus on lower elevation vegetation in Montana, below about 6,000 ft (1,830 m) or below montane forests dominated, for example, by *Pinus contorta*, *Picea engelmannii*, or *Pseudotsuga menziesii*.

The Grasses of Montana

The species enumerated in the Grasses of Montana (Lavin and Seibert 2009) were compared against the grasses listed for the adjacent plains state of South Dakota and for the more distant Intermountain state of Utah. The grass distribution data taken from the PLANTS database (USDA, NRCS 2008) was essentially derived from Cronquist and others (1972), McGregor and others (1977), VanBruggen (1985), and Welch and others (1987) and these works were consulted to validate species distributions. South Dakota and Utah comparisons were made in order to bias in favor of detecting a Great Plains influence on the flora and vegetation of eastern Montana. Because of its upland Black Hills country, South Dakota shares many western montane grasses with Montana (for example *Festuca idahoensis* and *Trisetum spicatum*) in addition to plains species shared with Montana but not Utah (for example *Muhlenbergia cuspidata*). Choosing Utah because of its Intermountain status also had the additional advantage of biasing against the detection of a predominant Intermountain influence on the flora and vegetation of Montana because Utah is much more southerly in latitude and not adjacent to Montana, in contrast to South Dakota. Comparing Montana with an adjacent state like Idaho or Wyoming would have biased the results in favor of detecting a strong Intermountain influence.

North American Geographical Distributions of Low Elevation Dominants

All native species known to characterize vegetation types across Montana and low elevations were inspected for Montana county distributions as well as North American distribution patterns. The MONT and MONTU herbaria and the PLANTS database served as the primary sources for information on geographic distribution.

Nomenclature

Binomial nomenclature followed primarily Dorn (1984) with the exception of certain grass genera where Lavin and Seibert (2009) was used. The PLANTS Database (USDA, NRCS 2008) was used only as a secondary source of binomial nomenclature because this database is prone to the uncritical adoption of any recent nomenclature. Regardless, all Latin binomials reported in this study can be entered into the PLANTS database for information on taxonomic authorship as well as nomenclatural and taxonomic synonymy.

Table 1—The 52 native species that have a primarily Great Plains distribution and that are restricted, or nearly so, to low elevation open dry settings in the eastern two-thirds of Montana. The species “reported from Montana” (Dorn 1984) have not been verified with herbarium specimens as occurring in the state. The azonal species inhabit primarily gumbo clay or rocky outcrops with sparse plant cover. The widely distributed North American grasses *Andropogon gerardii*, *Bouteloua dactyloides*, *Muhlenbergia cuspidata*, and *Panicum virgatum* are included in this list because they characterize in part Great Plains vegetation. An asterisk indicates those species represented by more than 10 collections in the holdings of MONT and MONTU.

Great Plains Species	Occurring in Intermountain Flora	Great Plains Species	Occurring in Intermountain Flora
Monocots		Dicots (continued)	
<i>Andropogon gerardii</i> *	Yes	<i>Evax prolifera</i>	No
<i>Andropogon hallii</i> *	Yes	<i>Evolvulus nuttallianus</i>	Yes
<i>Bouteloua dactyloides</i> *	Yes	<i>Happlopappus armerioides</i> *	Yes
<i>Bouteloua hirsute</i> ^a	Yes	<i>Happlopappus multicaulis</i>	No
<i>Carex inops</i> ssp. <i>heliophila</i> *	Yes	<i>Heuchera richardsonii</i> (azonal)	No
<i>Dichanthelium wilcoxianum</i>	No	<i>Ipomoea leptophylla</i>	No
<i>Muhlenbergia cuspidata</i> *	No	<i>Lesquerella arenosa</i>	No
<i>Panicum virgatum</i> *	Yes	<i>Linum rigidum</i>	No
<i>Stipa spartea</i>	No	<i>Mertensia lanceolata</i> *	Yes
<i>Yucca glauca</i> *	No	<i>Opuntia macrorhiza</i> ^a	Yes
Dicots		<i>Oxytropis lambertii</i> *	Yes
<i>Amorpha canescens</i>	No	<i>Penstemon albidus</i> *	No
<i>Artemisia longifolia</i> * (azonal)	No	<i>Penstemon angustifolius</i>	Yes
<i>Asclepias stenophylla</i>	No	<i>Penstemon grandiflorus</i>	No
<i>Astragalus plattensis</i>	No	<i>Phlox andicola</i>	No
<i>Astragalus racemosus</i>	Yes	<i>Physaria brassicoides</i>	No
<i>Castilleja sessiliflora</i> *	No	<i>Polygala alba</i> *	No
<i>Dalea candida</i> *	Yes	<i>Psoralea argophylla</i> *	No
<i>Dalea enneandra</i>	No	<i>Psoralea cuspidata</i> ^a	No
<i>Dalea purpurea</i> *	No	<i>Psoralea esculenta</i> *	No
<i>Dalea villosa</i>	No	<i>Psoralea hypogaea</i>	No
<i>Erigeron bellidiastrium</i>	Yes	<i>Psoralea tenuiflora</i>	Yes
<i>Eriogonum annuum</i> *	No	<i>Suckleya vermiculatus</i>	No
<i>Eriogonum pauciflorum</i> * (azonal)	No	<i>Tradescantia bracteata</i>	No
<i>Euphorbia missurica</i>	No	<i>Tradescantia occidentalis</i> *	Yes
<i>Eustoma grandiflorum</i>	No	<i>Triodanis leptocarpa</i>	No
		<i>Viola pedatifida</i> ^a	No

^aReported From Montana

RESULTS

Great Plains Species Occurring in Eastern Montana

Of the 2,500+ species of vascular plants in Montana, 52 native species that are generally confined to the North American Great Plains biome are represented in the eastern Montana flora (table 1, figures 1-3). Seventeen of these species are represented in the Intermountain flora. The large majority of these species, 42, are dicot forbs, whereas the remaining 10 are monocots. Of the monocots, the most abundant and widely distributed is *Yucca glauca*. With the exception of the short statured *Muhlenbergia cuspidata*, there are no distinctively Great Plains grasses that are abundant in eastern Montana, as for example, *Stipa spartea*. At least three of these Great Plains species, *Artemisia*

longifolia, *Eriogonum pauciflorum*, and *Heuchera richardsonii*, are azonal and thus not characteristic of vegetation with abundant grass cover. Only six of these 52 species are indicators of habitat types (*Andropogon gerardii*, *A. hallii*, *Carex inops*, *Panicum virgatum*, *Yucca glauca*, and *Artemisia longifolia*) that are representative of Great Plains vegetation (table 2). In general, herbarium specimens and field observations of most of these 52 species within Montana are few, if any, especially when compared to the many and more abundant species that ostensibly define Intermountain vegetation types (for example, species of *Agropyron* and *Artemisia*; table 2). Three of these 52, *Opuntia macrorhiza*, *Psoralea cuspidata*, and *Viola pedatifida* (figures 2-3) have been only reported to occur in Montana.

Eastern Montana Vegetation Types Dominated by Great Plains Species

Formally recognized vegetation types from open dry sites and that are suggestive of a Great Plains influence in eastern Montana are few and rare, in contrast to the formally recognized open dry-site vegetation types more characteristic of the Intermountain region, which are many and some widely distributed (table 2). Vegetation types in eastern Montana that are dominated by the tall warm season grasses *Andropogon gerardii*, *A. hallii*, and *Panicum virgatum*, as well as the cool season *Stipa spartea*, are non-existent, imperiled, or vulnerable (table 2).

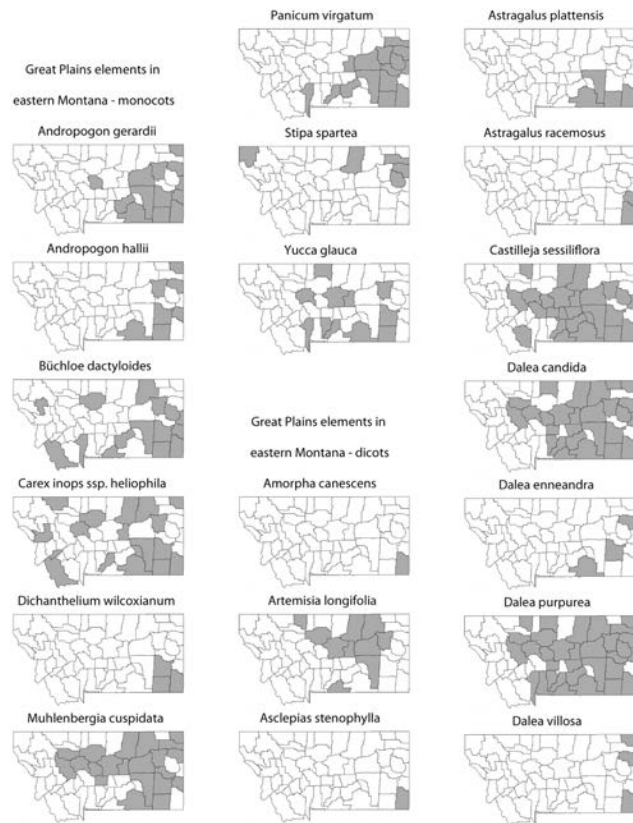


Figure 1—Great Plains elements in eastern Montana. Monocots: *Andropogon gerardii* – *Yucca glauca*. Dicots: *Amorphia canescens* – *Dalea villosa*.

The Grasses of Montana

Grass species occurring in Montana share more in common with the non-contiguous southerly state of Utah than with the adjacent plains state of South Dakota (table 3). The 207 grass species that Montana shares with Utah compare to 163 shared with South Dakota. This might be explained by the richness of certain mountain-inhabiting genera such as *Agrostis* and *Poa*. However, inspection of those species shared between just Montana and Utah reveals many low elevation grass species including both native and introduced. Most importantly, the grass species shared between just Montana and South Dakota are mostly widespread species. Only three of these, *Dichanthelium*

wilcoxianum, *Muhlenbergia cuspidata*, and *Stipa spartea*, have a truly Great Plains distribution within North America. *Dichanthelium wilcoxianum* and *Stipa spartea* are rare in Montana and are known from few specimens and relatively few counties (figure 1).

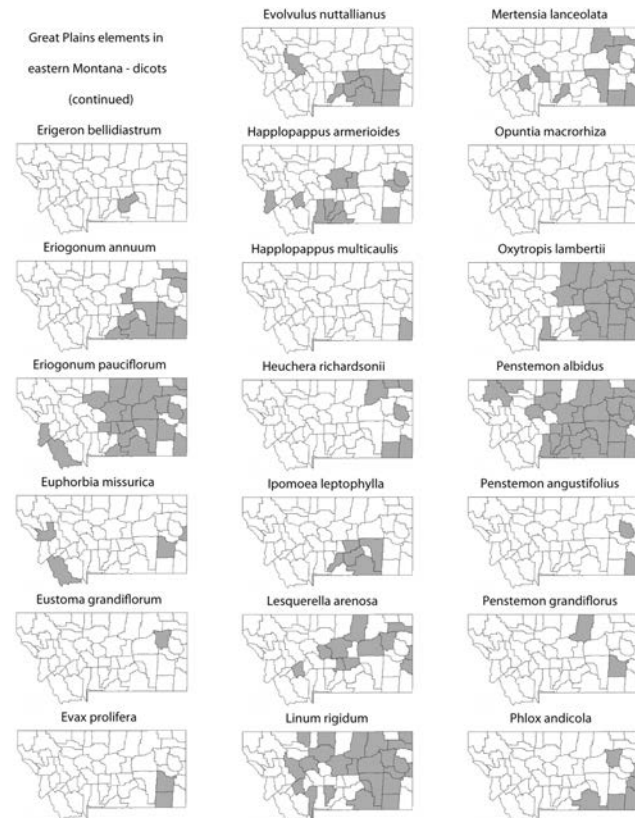


Figure 2—Great Plains dicot species with a distribution mainly or entirely in eastern Montana. The distribution map of *Opuntia macrorhiza* with no indication of a county occurrence signifies that this species is only reported to occur in eastern Montana (Dorn 1984).

North American Geographical Distributions of Low Elevation Dominants

Inspection of North American distributions of common low elevation and eastern Montana species revealed a Pacific Northwest, Boreal, Eastern Deciduous Forest, or Intermountain influence on the low elevation flora and vegetation. Individually and collectively these influences are perhaps more pronounced than the one from the Great Plains. A potentially significant Pacific Northwest influence on the lower elevation flora of Montana is possible and is exemplified by *Lysichiton americanus* and *Haplopappus carthamoides*. Such lower elevation species are found, however, in just northwestern Montana. Many species restricted in Montana to the northwestern portion of the state often had a very widespread North American distribution and no species with a primarily low elevation eastern Montana distribution was found to have a North American distribution that was exclusively from the Pacific Northwest.

A potential Boreal influence on the lower elevation flora of Montana is possible. This is exemplified by *Asclepias ovalifolia* and *Senecio congestus*, which occur in northcentral or northeastern Montana and more commonly at latitudes to the north throughout Canada (figure 3). This pattern was not detected commonly, however. An influence of the Eastern Deciduous Forests predominates in Montana along riparian or gallery forests, and is exemplified by *Fraxinus pennsylvanica*, *Populus deltoides*, *Quercus macrocarpa*, and *Ulmus americanus* (figures 3-4). These species occur as a western extension of eastern deciduous forests across the Great Plains. Common species across North America at low elevation settings may also have a predilection for eastern Montana, which is the case for *Acer negundo* and *Salix amygdaloides* (figure 4).

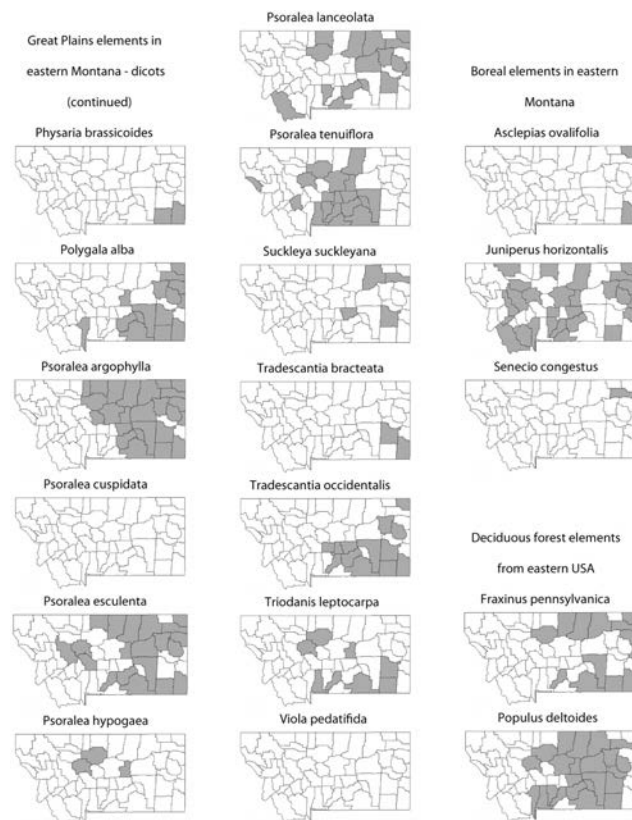


Figure 3—Distribution patterns of species occurring primarily in eastern Montana. Great Plains dicot species mainly or entirely in eastern Montana: *Physaria brassicoides* – *Viola pedatifida*. Examples of species with a primarily boreal distribution that otherwise occur in eastern Montana: *Asclepias ovalifolia* – *Senecio congestus*. Examples of species from eastern deciduous forests that occur natively in eastern Montana: *Fraxinus pennsylvanica* – *Populus deltoides*. The distribution maps of *Psoralea cuspidata* and *Viola pedatifida* with no indication of a county occurrence signifies that the corresponding species is only reported to occur in eastern Montana (Dorn 1984).

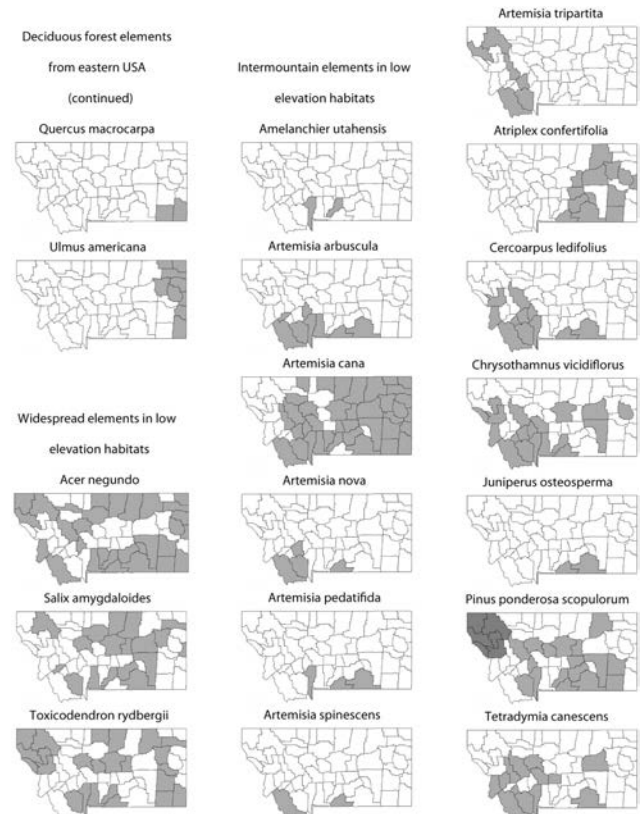


Figure 4—Distribution patterns of species occurring primarily in eastern Montana. Examples of species from eastern deciduous forests that occur natively in eastern Montana: *Quercus macrocarpa*–*Ulmus americana*. Examples of widespread North American species with a predilection for low elevation habitats especially in eastern Montana: *Acer negundo*–*Toxicodendron rydbergii*. Examples of species centered in the Intermountain region that have a predilection for low elevation habitats in Montana: *Amelanchier utahensis*–*Tetradymia canescens*. The county distribution of *Pinus ponderosa* var. *scopulorum* lies to the east of the continental divide, whereas the county distribution of var. *ponderosa* (darker gray) lies to the northwest part. Genetic and fossil evidence suggest that var. *scopulorum* migrated northward independently of var. *ponderosa* from a region including southwestern United States (Latta and Mitton 1999).

Perhaps one of the largest influences on the low elevation flora and vegetation of Montana, especially the eastern two-thirds of the state, comes from the Intermountain biome or from western North America in general. An Intermountain influence is readily exemplified by *Amelanchier utahensis*, *Artemisia arbuscula*, *Artemisia nova*, *Artemisia pedatifida*, *Artemisia spinescens*, *Artemisia tripartita*, *Cercocarpus ledifolius*, *Chrysothamnus viscidiflorus*, *Juniperus osteosperma*, *Tetradymia canescens*, and *Tetradymia spinosa* (figures 4-5). These species are found mostly in southwestern Montana and otherwise are concentrated in the Intermountain region. Notably, the two counties in the southwestern quadrant of Montana, Beaverhead and Carbon

counties (for example the two counties recorded for *A. spinescens*; figure 4) harbor many of these Intermountain elements. In addition, vegetation dominants that are widespread in western North American and that are also widespread at low elevations across Montana are exemplified by many species, especially grasses, and includes for example *Agropyron dasystachyum* and *Agropyron spicatum* (figure 5).

DISCUSSION

The four lines of evidence brought to bear on the question - to what degree does the Great Plains biome influence the flora and vegetation of eastern Montana - all point to a much greater influence of the Intermountain biome on this region. The foothills-prairie topography in Montana has little bearing on truly Great Plains floristic elements. Many of 52 Great Plains species are not common in Montana and three of these have only been reported from the state (figures 2-3). In addition, many of these 52 species belong to genera such as *Dalea*, *Eriogonum*, *Oxytropis*, *Penstemon*, and *Psoralea* that are well represented in open dry-site vegetation in the Intermountain flora (Cronquist and others 1972). Thus, the representation of these genera in the sagebrush steppe of eastern Montana is not necessarily an indication of Great Plains influence but rather the ecological predilection of such genera for open dry-site vegetation.

The finding that the low elevation dry-site vegetation in eastern Montana may be more influenced by the Intermountain than Great Plains biome is in accord with the difficulty of using the Great Plains Flora (McGregor and Barkley 1986) for plant identification in eastern Montana. The main issue is that the Great Plains Flora treats many plant taxa that are either not present, rare, or not diverse in Montana. Examples of taxa at the family level that are taxonomically diverse and abundant in the Great Plains flora but absent, rare, or not diverse in eastern Montana, include *Agavaceae*, *Alismataceae*, *Asclepidaceae*, *Caesalpinaceae*, *Cistaceae*, *Convolvulaceae*, *Euphorbiaceae*, *Fagaceae*, *Juglandaceae*, *Lauraceae*, *Linaceae*, *Lythraceae*, *Malvaceae*, *Moraceae*, *Mimosaceae*, *Nyctaginaceae*, *Platanaceae*, *Polygalaceae*, *Rubiaceae*, *Saururaceae*, *Smilacaceae*, *Solanaceae*, *Sparganiaceae*, and *Vitaceae*. Examples of this at the genus level include *Cyperus* and *Rhynchospora* (Cyperaceae), *Dalea*, *Desmodium*, and *Lespedeza* (Fabaceae), *Dichanthelium*, *Eragrostis*, *Muhlenbergia*, *Panicum*, and *Sporobolus* (Poaceae), and *Silphium* and *Vernonia* (Asteraceae). Such marked floristic distinctions suggest eastern Montana is not part of the Great Plains biome or metacommunity. That is, immigrants from the Great Plains biome into eastern Montana are not likely to successfully establish and proliferate, which is an important aspect of the metacommunity definition of Hubbell (2001).



Figure 5—Examples of species centered in the Intermountain region that have a predilection for low elevation habitats in Montana: *Tetradytmia spinosa*. Examples of vegetationally dominant western North American species that also dominate low elevation habitats from western Montana across the state to the very eastern tier of counties: *Agropyron dasystachyum* – *Shepherdia argentea*. The distribution of *Artemisia tridentata* reflects the distribution of the three main constituent subspecies, especially ssp. *tridentata* and ssp. *wyomingensis* across the state. This is also true for *Chrysothamnus nauseosus* and its two main constituents ssp. *nauseosus* and ssp. *consimilis*.

The biome maps presented in every flora and vegetation analysis involving the state of Montana (for example Dorn 1984; Mueggler and Stewart 1980) have unequivocally implicated the eastern portion of the state as being part of the Great Plains biome. According to the floristic evidence put forth in this study, the open shrub-grass dominated dry-site flora and vegetation in eastern Montana probably should not be managed as if they are Great Plains grasslands if only because they lack the abundance of productive warm season tall grass species. Indeed, the prevalence and diversity of short statured cool season grasses along with shrubby *Asteraceae* and *Chenopodiaceae* suggest that the open dry-site flora and vegetation of eastern Montana should be managed as if they are part of the Intermountain biome or metacommunity. All the evidence points to Intermountain and western North American plant species having an excellent ability immigrate into and proliferate throughout eastern Montana,

as exemplified by dominant plant species belonging to genera such as *Agropyron*, *Artemisia*, and *Atriplex*.

It is commonly asserted by APF that if the Lewis and Clark explorers had made an observation of herds of many thousands of bison in eastern Montana during the early 1800s (for example journal entry of Wednesday, May 8th, 1805; University of Nebraska Press 2005), then a benchmark for restoration of APF lands in eastern Montana should include large herds of bison in places like the Charles M. Russell National Wildlife Refuge in northcentral and eastern Montana. This benchmark not only disregards evidence that bison distributions were strongly influenced by Native Americans (for example Fisher and

Roll 1998), but it also uncritically accepts the conventional view that eastern Montana harbors Great Plains vegetation. The motivation for our study was to put forth the evidence that strongly questions this conventional view of the flora and vegetation of eastern Montana and to point to its strong Intermountain floristic affinities.

ACKNOWLEDGEMENTS

We thank Peter Lesica at the University of Montana Herbarium for facilitating access to the plant collections there. We also thank Tad Weaver for sharing his extensive insights into the vegetation of Montana.

Table 2—Low elevation dry-site vegetation types reported for the State of Montana (Montana Natural Heritage Program, 2002) that are characteristic of either Great Plains (e.g., Sims and Risser 2000) or Intermountain Provinces (e.g., West and Young 2000). State ranks: 1=critically imperiled, 2=imperiled, 3=vulnerable, 4=apparently secure, 5=demonstrably secure, ?=rank uncertain, Q=information needed.

Vegetation Types Characteristic of the Great Plains	Heritage Program State Rank
<i>Andropogon gerardii</i> - <i>Schizachyrium scoparium</i> Western Great Plains Herbaceous Vegetation	2
<i>Andropogon hallii</i> - <i>Calamovilfa longifolia</i> Herbaceous Vegetation	?
<i>Andropogon hallii</i> - <i>Carex inops</i> ssp. <i>heliophila</i> Herbaceous Vegetation	3
<i>Panicum virgatum</i> - <i>Muhlenbergia richardsonis</i> - (<i>Schizachyrium scoparium</i>) Herbaceous Vegetation	?
<i>Panicum virgatum</i> Herbaceous Vegetation (provisional)	2Q
<i>Schizachyrium scoparium</i> - <i>Bouteloua (curtipendula, gracilis)</i> - <i>Carex filifolia</i> Herbaceous Vegetation	?
<i>Schizachyrium scoparium</i> - <i>Carex inops</i> ssp. <i>heliophila</i> Herbaceous Vegetation	3
<i>Schizachyrium scoparium</i> - <i>Muhlenbergia cuspidata</i> Herbaceous Vegetation	2
<i>Yucca glauca</i> - <i>Calamovilfa longifolia</i> Shrub Herbaceous Vegetation	4
<i>Yucca glauca</i> - <i>Pseudoroegneria spicata</i> Shrub Herbaceous Vegetation	4Q
<i>Artemisia longifolia</i> - <i>Calamovilfa longifolia</i> Sparse Vegetation	3-4
<i>Artemisia longifolia</i> Badlands Sparse Vegetation	3-4
Vegetation Types Characteristic of the Intermountain Region	
<i>Amelanchier alnifolia</i> - <i>Pseudoroegneria spicata</i> Shrubland	2?
<i>Artemisia arbuscula</i> - <i>Festuca idahoensis</i> Dwarf Shrub Herbaceous Vegetation	3
<i>Artemisia arbuscula</i> - <i>Pseudoroegneria spicata</i> Dwarf Shrub Herbaceous Vegetation	3
<i>Artemisia arbuscula</i> ssp. <i>longiloba</i> - <i>Elymus lanceolatus</i> Dwarf Shrub Herbaceous Vegetation	?
<i>Artemisia arbuscula</i> ssp. <i>longiloba</i> - <i>Festuca idahoensis</i> Dwarf Shrub Herbaceous Vegetation	2
<i>Artemisia arbuscula</i> ssp. <i>longiloba</i> - <i>Pseudoroegneria spicata</i> Dwarf Shrub Herbaceous Vegetation	?
<i>Artemisia cana</i> - <i>Elymus lanceolatus</i> - <i>Nassella viridula</i> Shrub Herbaceous Vegetation	2-3?
<i>Artemisia cana</i> - <i>Elymus trachycaulus</i> Shrub Herbaceous Vegetation	?
<i>Artemisia cana</i> - <i>Festuca idahoensis</i> Shrub Herbaceous Vegetation	3?
<i>Artemisia cana</i> - <i>Leymus cinereus</i> Shrubland	?
<i>Artemisia cana</i> - <i>Pascopyrum smithii</i> Shrubland	4
<i>Artemisia cana</i> - <i>Poa pratensis</i> Shrub Herbaceous Vegetation	?
<i>Artemisia cana</i> - <i>Stipa comata</i> Shrub Herbaceous Vegetation	3
<i>Artemisia cana</i> ssp. <i>cana</i> - <i>Pascopyrum smithii</i> Shrub Herbaceous Vegetation	?
<i>Artemisia cana</i> ssp. <i>viscidula</i> - <i>Festuca idahoensis</i> Shrubland	?
<i>Artemisia nova</i> - <i>Festuca idahoensis</i> Dwarf Shrub Herbaceous Vegetation	1-2
<i>Artemisia nova</i> - <i>Pseudoroegneria spicata</i> Dwarf Shrubland	3
<i>Artemisia pedatifida</i> - <i>Atriplex gardneri</i> Dwarf Shrubland	2
<i>Artemisia pedatifida</i> - <i>Festuca idahoensis</i> Dwarf Shrub Herbaceous Vegetation	2
<i>Artemisia pedatifida</i> - <i>Pseudoroegneria spicata</i> Dwarf Shrub Herbaceous Vegetation	3

Table—2 (Continued)	
Vegetation Types Characteristic of the Intermountain Region	Heritage Program State Rank
<i>Artemisia spinescens</i> Dwarf Shrubland	?
<i>Artemisia tridentata</i> - <i>Atriplex confertifolia</i> Shrubland	4
<i>Artemisia tridentata</i> - <i>Festuca campestris</i> Shrub Herbaceous Vegetation	3
<i>Artemisia tridentata</i> - <i>Festuca idahoensis</i> Shrub Herbaceous Vegetation	4?
<i>Artemisia tridentata</i> - <i>Stipa comata</i> Shrubland	4
<i>Artemisia tridentata</i> ssp. <i>tridentata</i> - <i>Festuca idahoensis</i> Shrubland	?
<i>Artemisia tridentata</i> ssp. <i>tridentata</i> - <i>Leymus cinereus</i> Shrubland	1
<i>Artemisia tridentata</i> ssp. <i>tridentata</i> - <i>Pascopyrum smithii</i> Shrubland	3
<i>Artemisia tridentata</i> ssp. <i>tridentata</i> - <i>Pseudoroegneria spicata</i> Shrub Herbaceous Vegetation	?
<i>Artemisia tridentata</i> ssp. <i>tridentata</i> - <i>Stipa comata</i> Shrub Herbaceous Vegetation	?
<i>Artemisia tridentata</i> ssp. <i>vaseyana</i> - <i>Bromus carinatus</i> Shrubland	?
<i>Artemisia tridentata</i> ssp. <i>vaseyana</i> - <i>Festuca campestris</i> Shrub Herbaceous Vegetation	?
<i>Artemisia tridentata</i> ssp. <i>vaseyana</i> - <i>Festuca idahoensis</i> - <i>Elymus trachycaulus</i> Shrubland	?
<i>Artemisia tridentata</i> ssp. <i>vaseyana</i> - <i>Festuca idahoensis</i> Shrub Herbaceous Vegetation	4
<i>Artemisia tridentata</i> ssp. <i>vaseyana</i> - <i>Festuca idahoensis</i> - <i>Pascopyrum smithii</i> Shrubland	?
<i>Artemisia tridentata</i> ssp. <i>vaseyana</i> - <i>Festuca idahoensis</i> - <i>Pseudoroegneria spicata</i> Shrubland	?
<i>Artemisia tridentata</i> ssp. <i>vaseyana</i> - <i>Festuca idahoensis</i> - <i>Stipa richardsonii</i> Shrubland	?
<i>Artemisia tridentata</i> ssp. <i>vaseyana</i> - <i>Festuca idahoensis</i> - <i>Geranium viscosissimum</i> Shrubland	?
<i>Artemisia tridentata</i> ssp. <i>vaseyana</i> - <i>Festuca idahoensis</i> - <i>Stipa occidentalis</i> Shrubland	?
<i>Artemisia tridentata</i> ssp. <i>vaseyana</i> - <i>Leymus cinereus</i> Shrubland	?
<i>Artemisia tridentata</i> ssp. <i>vaseyana</i> - <i>Pascopyrum smithii</i> Shrubland	3?
<i>Artemisia tridentata</i> ssp. <i>vaseyana</i> - <i>Poa pratensis</i> Shrub Herbaceous Vegetation	?
<i>Artemisia tridentata</i> ssp. <i>vaseyana</i> - <i>Pseudoroegneria spicata</i> Shrubland	?
<i>Artemisia tridentata</i> ssp. <i>vaseyana</i> - <i>Symphoricarpos oreophilus</i> - <i>Festuca idahoensis</i> Shrubland	?
<i>Artemisia tridentata</i> ssp. <i>wyomingensis</i> - <i>Atriplex confertifolia</i> Shrubland	3
<i>Artemisia tridentata</i> ssp. <i>wyomingensis</i> - <i>Bouteloua gracilis</i> Shrubland	?
<i>Artemisia tridentata</i> ssp. <i>wyomingensis</i> - <i>Carex filifolia</i> Shrubland	?
<i>Artemisia tridentata</i> ssp. <i>wyomingensis</i> - <i>Elymus lanceolatus</i> ssp. <i>albicans</i> Shrubland	4?
<i>Artemisia tridentata</i> ssp. <i>wyomingensis</i> - <i>Pascopyrum smithii</i> Shrub Herbaceous Vegetation	?
<i>Artemisia tridentata</i> ssp. <i>wyomingensis</i> - <i>Poa secunda</i> Shrubland	?
<i>Artemisia tridentata</i> ssp. <i>wyomingensis</i> - <i>Pseudoroegneria spicata</i> Shrub Herbaceous Vegetation	3
<i>Artemisia tridentata</i> ssp. <i>wyomingensis</i> - <i>Pseudoroegneria spicata</i> Shrubland	5?
<i>Artemisia tripartita</i> - <i>Elymus lanceolatus</i> Shrub Herbaceous Vegetation	?
<i>Artemisia tripartita</i> - <i>Festuca campestris</i> Shrub Herbaceous Vegetation	2
<i>Artemisia tripartita</i> - <i>Festuca idahoensis</i> Shrub Herbaceous Vegetation	3?
<i>Artemisia tripartita</i> - <i>Pseudoroegneria spicata</i> Shrub Herbaceous Vegetation	?
<i>Artemisia tripartita</i> ssp. <i>tripartita</i> - <i>Hesperostipa comata</i> Shrub Herbaceous Vegetation	?
<i>Atriplex confertifolia</i> - <i>Chrysothamnus nauseosus</i> Shrubland	3Q
<i>Atriplex confertifolia</i> Wyoming Basins Shrubland	1-2
<i>Atriplex gardneri</i> - <i>Artemisia spinescens</i> Dwarf Shrubland	1
<i>Atriplex gardneri</i> - <i>Artemisia tridentata</i> Dwarf Shrubland	3
<i>Atriplex gardneri</i> - <i>Endolepis dioica</i> Dwarf Shrubland	2?
<i>Atriplex gardneri</i> - <i>Monolepis nuttalliana</i> Dwarf Shrubland	2
<i>Atriplex gardneri</i> - <i>Oryzopsis hymenoides</i> Dwarf Shrubland	3
<i>Atriplex gardneri</i> - <i>Pascopyrum smithii</i> Dwarf Shrubland	3
<i>Atriplex gardneri</i> Dwarf Shrubland	2-3
<i>Krascheninnikovia lanata</i> - <i>Phlox</i> spp. Dwarf Shrubland	3Q
<i>Krascheninnikovia lanata</i> - <i>Stipa comata</i> Dwarf Shrubland	3
<i>Rhus trilobata</i> - <i>Calamovilfa longifolia</i> Shrub Herbaceous Vegetation	3Q
<i>Rhus trilobata</i> - <i>Carex filifolia</i> Shrub Herbaceous Vegetation	3

Table—2 (Continued)	
Vegetation Types Characteristic of the Intermountain Region	Heritage Program State Rank
<i>Rhus trilobata</i> - <i>Elymus lanceolatus</i> - <i>Nassella viridula</i> Shrub Herbaceous Vegetation	2-3?
<i>Rhus trilobata</i> - <i>Festuca idahoensis</i> Shrub Herbaceous Vegetation	2?
<i>Rhus trilobata</i> - <i>Pseudoroegneria spicata</i> Shrub Herbaceous Vegetation	4
<i>Rhus trilobata</i> - <i>Schizachyrium scoparium</i> Shrub Herbaceous Vegetation	3
<i>Sarcobatus vermiculatus</i> - <i>Artemisia tridentata</i> Shrubland	4
<i>Sarcobatus vermiculatus</i> - <i>Atriplex gardneri</i> Shrubland	3
<i>Sarcobatus vermiculatus</i> - <i>Distichlis spicata</i> Shrubland	2
<i>Sarcobatus vermiculatus</i> - <i>Leymus cinereus</i> Shrubland	2-3
<i>Sarcobatus vermiculatus</i> - <i>Pascopyrum smithii</i> Shrub Herbaceous Vegetation	4
<i>Sarcobatus vermiculatus</i> - <i>Pseudoroegneria spicata</i> Shrubland	3

Table 3—Poaceae in Utah (UT), Montana (MT), and South Dakota (SD) including native and introduced taxa. 0 = species absent in the state, 1 = species present in the state. Data taken from PLANTS database (USDA, NRCS, 2008) with nomenclature modified from Lavin and Seibert (2009). Utah and Montana share 207 species, Utah and South Dakota share 150, and Montana and South Dakota share 163 species. The “shared” column indicates those species shared by just two of the states. The asterisks (with line in boldface) associated with a MT-SD shared species indicate only three grass species with a truly Great Plains distribution that includes eastern Montana and South Dakota but not Utah (*Dichanthelium wilcoxianum*, *Muhlenbergia cuspidata*, and *Stipa spartea*).

Species	UT	MT	SD	Shared	Species	UT	MT	SD	Shared
<i>Aegilops cylindrica</i>	1	1	1		<i>Agropyron albicans</i>	1	1	1	
<i>Agropyron cristatum</i>	1	1	1		<i>Agropyron dasystachyum</i>	1	1	1	
<i>Agropyron elongatum</i>	1	1	1		<i>Agropyron fragile</i>	1	0	1	UT-SD
<i>Agropyron intermedium</i>	1	1	1		<i>Agropyron pseudorepens</i>	1	0	0	
<i>Agropyron repens</i>	1	1	1		<i>Agropyron scribneri</i>	1	1	0	UT-MT
<i>Agropyron smithii</i>	1	1	1		<i>Agropyron spicatum</i>	1	1	1	
<i>Agropyron trachycaulum</i>	1	1	1		<i>Agropyron trichophorum</i>	1	1	0	UT-MT
<i>Agrostis borealis</i>	0	1	0		<i>Agrostis diegoensis</i>	1	1	0	
<i>Agrostis exarata</i>	1	1	1		<i>Agrostis humilis</i>	1	1	0	UT-MT
<i>Agrostis hyemalis</i>	0	0	1		<i>Agrostis idahoensis</i>	1	1	0	UT-MT
<i>Agrostis oregonensis</i>	0	1	0		<i>Agrostis perennans</i>	0	0	1	
<i>Agrostis rossiae</i>	0	1	0		<i>Agrostis scabra</i>	1	1	0	UT-MT
<i>Agrostis semiverticillata</i>	1	0	0		<i>Agrostis stolonifera</i>	1	1	1	
<i>Agrostis tenuis</i>	1	1	0	UT-MT	<i>Agrostis thurberiana</i>	1	1	0	UT-MT
<i>Agrostis variabilis</i>	1	1	0	UT-MT	<i>Alopecurus aequalis</i>	1	1	1	
<i>Alopecurus alpinus</i>	1	1	0	UT-MT	<i>Alopecurus</i>				
<i>Alopecurus carolinianus</i>	1	1	1		<i>arundinaceus</i>	0	1	0	
<i>Alopecurus pratensis</i>	1	1	0	UT-MT	<i>Alopecurus geniculatus</i>	1	1	1	
<i>Andropogon hallii</i>	1	1	1		<i>Andropogon gerardii</i>	1	1	1	
<i>Andropogon scoparium</i>	1	1	1		<i>Andropogon barbinodis</i>	1	0	0	
<i>Aristida adscensionis</i>	1	0	0		<i>Andropogon ischaemum</i>	1	1	0	UT-MT
<i>Aristida longispica</i>	0	0	1		<i>Apera interrupta</i>	1	1	0	
<i>Aristida purpurea</i>					<i>Aristida dichotoma</i>	0	0	1	
<i>longiseta</i>	1	1	1		<i>Aristida oligantha</i>	0	0	1	
<i>Avena barbata</i>	0	1	0		<i>Arrhenatherum elatius</i>	1	1	1	
<i>Avena sativa</i>	1	1	1		<i>Avena fatua</i>	1	1	1	
<i>Blepharidachne kingii</i>	1	0	0		<i>Beckmannia syzigachne</i>	1	1	1	
<i>Bouteloua barbata</i>	1	1	0		<i>Blepharoneuron</i>				
<i>Bouteloua eripoda</i>	1	0	0		<i>tricholepis</i>	1	0	0	
<i>Bouteloua hirsuta</i>	1	1	1		<i>Bouteloua curtipendula</i>	1	1	1	
<i>Bouteloua trifida</i>	1	0	0		<i>Bouteloua gracilis</i>	1	1	1	
<i>Bromus briziformis</i>	1	1	1		<i>Bouteloua simplex</i>	1	0	0	
<i>Bromus ciliatus</i>	1	1	1		<i>Bromus anomalus</i>	1	1	1	
<i>Bromus diandrus</i>	1	1	0	UT-MT	<i>Bromus carinatus</i>	1	1	1	
<i>Bromus japonicus</i>	1	1	1		<i>Bromus commutatus</i>	1	1	0	UT-MT
<i>Bromus latiglumis</i>	0	1	1	MT-SD	<i>Bromus inermis</i>	1	1	1	
					<i>Bromus kalmii</i>	0	0	1	

Table 3 (Continued)—Poaceae in Utah (UT), Montana (MT), and South Dakota (SD) including native and introduced taxa.

Species	UT	MT	SD	Shared	Species	UT	MT	SD	Shared
<i>Bromus pubescens</i>	0	1	1	MT-SD	<i>Bromus mollis</i>	1	1	1	
<i>Bromus rigidus</i>	1	0	0		<i>Bromus pumpellianus</i>	1	1	0	UT-MT
<i>Bromus secalinus</i>	1	1	1		<i>Bromus rubens</i>	1	0	0	
<i>Bromus sterilis</i>	1	0	0		<i>Bromus squarrosus</i>	0	1	1	MT-SD
<i>Bromus vulgaris</i>	1	1	0		<i>Bromus tectorum</i>	1	1	1	
<i>Buchloe dactyloides</i>	1	1	1		<i>Bromus willdenowii</i>	1	0	0	
<i>Calamagrostis koelerioides</i>	0	1	0		<i>Calamagrostis canadensis</i>	1	1	1	
<i>Calamagrostis purpurascens</i>	1	1	1		<i>Calamagrostis montanensis</i>	0	1	1	MT-SD
<i>Calamagrostis scopulorum</i>	1	1	0	UT-MT	<i>Calamagrostis rubescens</i>	1	1	0	UT-MT
<i>Calamagrostis tweedyi</i>	0	1	0		<i>Calamagrostis stricta</i>	1	1	1	
<i>Calamovilfa longifolia</i>	0	1	1	MT-SD	<i>Calamovilfa gigantea</i>	1	0	0	
<i>Cenchrus longispinus</i>	1	1	1		<i>Catabrosa aquatica</i>	1	1	1	
<i>Chloris virgata</i>	1	0	0		<i>Chloris verticillata</i>	0	0	1	
<i>Cinna latifolia</i>	1	1	1		<i>Cinna arundinacea</i>	0	1	1	MT-SD
<i>Crypsis schoenoides</i>	1	1	0	UT-MT	<i>Crypsis alopecuroides</i>	1	1	0	UT-MT
<i>Cynosurus cristatus</i>	0	1	0		<i>Cynodon dactylon</i>	1	1	0	UT-MT
<i>Dactylis glomerata</i>	1	1	1		<i>Cynosurus echinatus</i>	0	1	0	
<i>Danthonia intermedia</i>	1	1	1		<i>Danthonia californica</i>	1	1	1	
<i>Danthonia spicata</i>	0	1	1	MT-SD	<i>Danthonia parryi</i>	0	1	0	
<i>Deschampsia atropurpurea</i>	0	1	0		<i>Danthonia unispicata</i>	1	1	1	
<i>Deschampsia danthonioides</i>	1	1	0	UT-MT	<i>Deschampsia cespitosa</i>	1	1	0	UT-MT
<i>Diarrhena americana</i>	0	0	1		<i>Deschampsia elongata</i>	1	1	0	UT-MT
<i>Dichanthelium oligosanthes</i>	1	1	1		<i>Dichanthelium lanuginosum</i>	1	1	1	
<i>Dichanthelium xanthophyllum</i>	0	0	1		<i>Dichanthelium wilcoxianum</i>	0	1	1	MT-SD*
<i>Dichanthelium linearifolium</i>	0	0	1		<i>Dichanthelium leibergii</i>	0	0	1	
<i>Digitaria adscendens</i>	1	0	0		<i>Dichanthelium villosissimum</i>	0	0	1	
<i>Digitaria sanguinalis</i>	1	1	1		<i>Digitaria ischaemum</i>	1	1	1	
<i>Echinochloa crusgalli</i>	1	1	1		<i>Distichlis stricta</i>	1	1	1	
<i>Eleusine indica</i>	1	1	1		<i>Echinochloa muricata</i>	1	1	1	
<i>Elyhordeum montanense</i>	0	0	1		<i>Elyhordeum dakotense</i>	0	0	1	
<i>Elymus aristatus</i>	1	0	0		<i>Elymus ambiguus</i>	1	1	0	UT-MT
<i>Elymus canadensis</i>	1	1	1		<i>Elymus cinereus</i>	1	1	1	
<i>Elymus elymoides</i>	1	1	1		<i>Elymus diversiglumis</i>	0	0	1	
<i>Elymus glaucus</i>	1	1	1		<i>Elymus flavescens</i>	1	1	0	UT-MT
<i>Elymus hystrix</i>	0	0	1		<i>Elymus hansenii</i>	1	0	0	
<i>Elymus interruptus</i>	0	0	1		<i>Elymus innovatus</i>	0	1	1	MT-SD
<i>Elymus macounii</i>	1	1	1		<i>Elymus junceus</i>	1	1	1	
<i>Elymus saundersii</i>	1	0	0		<i>Elymus salina</i>	1	0	0	
<i>Elymus triticoides</i>	1	1	0	UT-MT	<i>Elymus simplex</i>	1	0	0	
<i>Elymus virginicus</i>	1	1	1		<i>Elymus villosus</i>	0	0	1	
<i>Eragrostis cilianensis</i>	1	1	1		<i>Enneapogon desvauxii</i>	1	0	0	
<i>Eragrostis hypnoides</i>	1	1	1		<i>Eragrostis curvula</i>	1	0	0	
<i>Eragrostis orcuttiana</i>	1	0	0		<i>Eragrostis minor</i>	1	1	1	
<i>Eragrostis pilosa</i>	0	0	1		<i>Eragrostis pectinacea</i>	1	1	1	
<i>Eragrostis spectabilis</i>	0	0	1		<i>Eragrostis reptans</i>	0	0	1	
<i>Eremopyrum triticeum</i>	1	1	0	UT-MT	<i>Eragrostis trichodes</i>	0	0	1	
<i>Festuca arizonica</i>	1	0	0		<i>Erioneuron pulchellum</i>	1	0	0	
<i>Festuca campestris</i>	0	1	0		<i>Festuca baffinensis</i>	1	1	0	UT-MT
<i>Festuca idahoensis</i>	1	1	1		<i>Festuca dasyclada</i>	1	0	0	
<i>Festuca obtusata</i>	0	0	1		<i>Festuca minutiflora</i>	1	1	0	UT-MT
<i>Festuca rubra</i>	1	1	0	UT-MT	<i>Festuca occidentalis</i>	1	1	1	
<i>Festuca subulata</i>	1	1	1		<i>Festuca sororia</i>	1	0	0	
<i>Festuca viridula</i>	0	1	0		<i>Festuca thurberi</i>	1	0	0	
<i>Glyceria borealis</i>	1	1	1		<i>Festuca viviparoidea</i>	0	1	0	

Table 3 (Continued)—Poaceae in Utah (UT), Montana (MT), and South Dakota (SD) including native and introduced taxa.

Species	UT	MT	SD	Shared	Species	UT	MT	SD	Shared
<i>Glyceria fluitans</i>	0	0	1		<i>Glyceria elata</i>	1	1	1	
<i>Glyceria striata</i>	1	1	1		<i>Glyceria grandis</i>	1	1	1	
<i>Helictotrichon</i>									
<i>mortonianum</i>	1	0	0		<i>Helictotrichon hookeri</i>	0	1	1	MT-SD
<i>Hilaria jamesii</i>	1	0	0		<i>Hierochloe odorata</i>	1	1	1	
<i>Holcus lanatus</i>	1	1	0	UT-MT	<i>Hilaria rigida</i>	1	0	0	
<i>Hordeum depressum</i>	1	0	0		<i>Hordeum brachyantherum</i>	1	1	0	UT-MT
<i>Hordeum glaucum</i>	1	0	0		<i>Hordeum geniculatum</i>	1	1	0	UT-MT
<i>Hordeum leporinum</i>	1	1	0		<i>Hordeum jubatum</i>	1	1	1	
<i>Hordeum vulgare</i>	1	1	1		<i>Hordeum pusillum</i>	1	1	1	
<i>Imperata brevifolia</i>	1	0	0		<i>Hystrix patula</i>	0	0	1	
<i>Leersia oryzoides</i>	1	1	1		<i>Koeleria nitida</i>	1	1	1	
<i>Leptochloa fusca</i>									
<i>fascicularis</i>	1	1	1		<i>Leersia virginica</i>	0	0	1	
<i>Leucopoa kingii</i>	1	1	1		<i>Leptochloa panicea</i>	1	0	1	UT-SD
<i>Lolium perenne</i>	1	1	1		<i>Lolium multiflorum</i>	1	1	0	UT-MT
<i>Lolium temulentum</i>	0	1	1	MT-SD	<i>Lolium persicum</i>	0	1	0	
<i>Melica smithii</i>	0	1	1	MT-SD	<i>Melica bulbosa</i>	1	1	1	
<i>Melica stricta</i>	1	0	0		<i>Melica spectabilis</i>	1	1	0	UT-MT
<i>Milium effusum</i>	0	0	1		<i>Melica subulata</i>	0	1	1	MT-SD
<i>Muhlenbergia andina</i>	1	1	0	UT-MT	<i>Monroa squarrosa</i>	1	1	1	
<i>Muhlenbergia asperifolia</i>	1	1	1		<i>Muhlenbergia arsenei</i>	1	0	0	
<i>Muhlenbergia filiculmnis</i>	1	0	0		<i>Muhlenbergia cuspidata</i>	0	1	1	MT-SD*
<i>Muhlenbergia frondosa</i>	0	0	1		<i>Muhlenbergia filiformis</i>	1	1	1	
<i>Muhlenbergia mexicana</i>	1	1	1		<i>Muhlenbergia glomerata</i>	1	1	1	
<i>Muhlenbergia montana</i>	1	1	0	UT-MT	<i>Muhlenbergia minutissima</i>	1	1	1	
<i>Muhlenbergia porteri</i>	1	0	0		<i>Muhlenbergia pauciflora</i>	1	0	0	
<i>Muhlenbergia racemosa</i>	1	1	1		<i>Muhlenbergia pungens</i>	1	0	1	UT-SD
<i>Muhlenbergia schreberi</i>	1	0	1	UT-SD	<i>Muhlenbergia richardsonis</i>	1	1	1	
<i>Muhlenbergia thurberi</i>	1	0	0		<i>Muhlenbergia sylavatica</i>	0	0	1	
<i>Oryzopsis asperifolia</i>	1	1	1		<i>Muhlenbergia wrightii</i>	1	0	0	
<i>Oryzopsis exigua</i>	1	1	0	UT-MT	<i>Oryzopsis contracta</i>	0	1	0	
<i>Oryzopsis micrantha</i>	1	1	1		<i>Oryzopsis hymenoides</i>	1	1	1	
<i>Oryzopsis pungens</i>	0	0	1		<i>Oryzopsis miliacea</i>	1	0	0	
<i>Oryzopsis webberi</i>	1	0	1	UT-MT	<i>Oryzopsis racemosa</i>	0	0	1	
<i>Panicum capillare</i>	1	1	1		<i>Panicum bulbosum</i>	1	0	0	
<i>Panicum flexile</i>	1	0	1		<i>Panicum dichotomiflorum</i>	1	1	1	
<i>Panicum miliaceum</i>	1	1	1		<i>Panicum hallii</i>	1	0	0	
<i>Panicum virgatum</i>	1	1	1		<i>Panicum obtusum</i>	1	0	0	
<i>Paspalum setaceum</i>	0	0	1		<i>Paspalum distichum</i>	1	0	0	
<i>Phalaris arundinacea</i>	1	1	1		<i>Pennisetum glaucum</i>	1	0	1	UT-SD
<i>Phippsia algida</i>	0	1	0		<i>Phalaris canariensis</i>	1	1	1	
<i>Phleum pratense</i>	1	1	1		<i>Phleum alpinum</i>	1	1	1	
<i>Piptantherum racemosum</i>	0	0	1		<i>Phragmites australis</i>	1	1	1	
<i>Poa annua</i>	1	1	1		<i>Poa alpina</i>	1	1	0	UT-MT
<i>Poa bigelovii</i>	1	0	0		<i>Poa arida</i>	1	1	1	
<i>Poa bulbosa</i>	1	1	1		<i>Poa bolanderi</i>	1	0	0	
<i>Poa compressa</i>	1	1	1		<i>Poa canbyi</i>	0	0	1	
<i>Poa cusickii</i>	1	1	0	UT-MT	<i>Poa curta</i>	1	0	0	
<i>Poa grayana</i>	1	1	0	UT-MT	<i>Poa fendleriana</i>	1	1	1	
<i>Poa juncifolia</i>	1	1	1		<i>Poa interior</i>	1	1	1	
<i>Poa leibergii</i>	1	0	0		<i>Poa leptocoma</i>	1	1	0	UT-MT
<i>Poa lettermanii</i>	1	1	0	UT-MT	<i>Poa palustris</i>	1	1	1	
<i>Poa pattersonii</i>	1	1	0	UT-MT	<i>Poa pratensis</i>	1	1	1	
<i>Poa reflexa</i>	1	1	0	UT-MT	<i>Poa rupicola</i>	1	1	0	UT-MT
<i>Poa secunda</i>	1	1	1		<i>Poa stenantha</i>	0	1	0	
<i>Poa suksdorfii</i>	1	0	0		<i>Poa sylvestris</i>	0	0	1	
<i>Poa trivialis</i>	1	1	1		<i>Poa wheeleri</i>	1	1	0	UT-MT
<i>Polypogon interruptus</i>	1	0	0		<i>Polypogon monspeliensis</i>	1	1	1	
<i>Puccinellia cusickii</i>	0	1	0		<i>Puccinellia distans</i>	1	1	1	
<i>Puccinellia fasciculata</i>	1	0	0		<i>Puccinellia lemmonii</i>	1	1	0	UT-MT
<i>Puccinellia nuttalliana</i>	1	1	1		<i>Puccinellia pauciflora</i>	1	1	1	
<i>Puccinellia simplex</i>	1	0	0		<i>Redfieldia flexuosa</i>	1	1	1	

Table 3 (Continued)—Poaceae in Utah (UT), Montana (MT), and South Dakota (SD) including native and introduced taxa.

Species	UT	MT	SD	Shared	Species	UT	MT	SD	Shared
<i>Schedonnardus paniculatus</i>	1	1	1		<i>Schedonorus arundinacea</i>	1	1	1	
<i>Schedonorus pratensis</i>	1	1	1		<i>Schismus barbatus</i>	1	0	0	
<i>Schizachne purpurascens</i>	1	1	1		<i>Sclerochloa dura</i>	1	1	0	UT-MT
<i>Scleropogon brevifolius</i>	1	0	0		<i>Scolochloa festucacea</i>	1	1	1	
<i>Secale cereale</i>	1	1	1		<i>Setaria faberi</i>	0	0	1	
<i>Setaria italica</i>	0	1	1	MT-SD	<i>Setaria pumila</i>	1	1	1	
<i>Setaria verticillata</i>	1	1	1		<i>Setaria viridis</i>	1	1	1	
<i>Sorghastrum nutans</i>	1	1	1		<i>Sorghum bicolor</i>	1	1	1	
<i>Sorghum halepense</i>	1	1	1		<i>Spartina gracilis</i>	1	1	1	
<i>Spartina pectinata</i>	1	1	1		<i>Sphenopholis obtusata</i>	1	1	1	
<i>Sporobolus aroides</i>	1	1	1		<i>Sporobolus compositus</i>	1	1	1	
<i>Sporobolus contractus</i>	1	0	0		<i>Sporobolus cryptandrus</i>	1	1	1	
<i>Sporobolus flexuosus</i>	1	0	0		<i>Sporobolus giganteus</i>	1	0	0	
<i>Sporobolus heterolepis</i>	0	1	1	MT-SD	<i>Sporobolus neglectus</i>	0	1	1	MT-SD
<i>Sporobolus pulvinatus</i>	1	0	0		<i>Sporobolus vaginiflorus</i>	0	0	1	
<i>Stipa arida</i>	1	0	0		<i>Stipa bloomeri</i>	1	0	0	
<i>Stipa comata</i>	1	1	1		<i>Stipa coronata</i>	1	0	0	
<i>Stipa lettermanii</i>	1	1	0	UT-MT	<i>Stipa nelsonii</i>	1	1	0	UT-MT
<i>Stipa neomexicana</i>	1	0	0		<i>Stipa nevadensis</i>	1	0	0	
<i>Stipa occidentalis</i>	1	1	1		<i>Stipa pinetorum</i>	1	1	0	UT-MT
<i>Stipa richardsonii</i>	0	1	1		<i>Stipa scribneri</i>	1	0	0	
<i>Stipa spartea</i>	0	1	1	MT-SD*	<i>Stipa speciosa</i>	1	0	0	
<i>Stipa thurberiana</i>	1	1	0	UT-MT	<i>Stipa viridula</i>	1	1	1	
<i>Stipa webberi</i>	1	0	0		<i>Stipa williamsii</i>	1	1	0	UT-MT
<i>Taeniatherum</i>					<i>Tridens muticus</i>	1	0	0	
<i>caputmedusae</i>	1	1	0	UT-MT	<i>Trisetum canescens</i>	1	1	0	UT-MT
<i>Triplasis purpurea</i>	0	0	1		<i>Trisetum orthochaetum</i>	0	1	0	
<i>Trisetum cernuum</i>	0	1	0		<i>Trisetum wolffii</i>	1	1	0	UT-MT
<i>Trisetum spicatum</i>	1	1	1		<i>Ventenata dubia</i>	0	1	0	
<i>Triticum aestivum</i>	1	1	1		<i>Vulpia microstachys</i>	1	1	0	UT-MT
<i>Vulpia bromoides</i>	1	1	0	UT-MT	<i>Vulpia octoflora</i>	1	1	1	
<i>Vulpia myuros</i>	1	1	0	UT-MT	<i>Zizania palustris</i>	0	1	1	MT-SD
<i>Zea mays</i>	1	1	1						

Total Poaceae species in each state: UT=284, MT=249, and SD=208

REFERENCES

- Cronquist, A., Holmgren, A.H., Holmgren, N.H., Reveal, J.L., Holmgren, P.K. 1972. Intermountain flora, volume one: geological and botanical history, plant geography, vascular cryptogams, gymnosperms, glossary. New York, NY: New York Botanical Garden Press. 270 p.
- Dorn, R.D. 1984. Vascular plants of Montana. Cheyenne, WY: Mountain West Publishing. 276 p.
- Fisher, J.W., Roll, T. 1998. Ecological relationships between bison and native Americans during late prehistory and the early historic period. Proceedings of the International Symposium on Bison Ecology and Management, pp. 283-302. 1997 June 4-7; Bozeman, MT.
- Hitchcock, C.L., Cronquist, A. 1973. Flora of the Pacific Northwest: an illustrated manual (the condensation of the original five volume work). Seattle, WA: University of Washington Press.
- Hubbell, S.P. 2001. The Unified Neutral Theory of Biodiversity and Biogeography. Princeton, NJ: Princeton University Press.
- Küchler, A.W. 1964. Potential Natural Vegetation of the Conterminous United States. New York, NY: American Geographical Society.
- Latta, R.G., Mitton, J.B. 1999. Historical separation and present gene flow through a zone of secondary contact in ponderosa pine. *Evolution* 53: 769-776.
- Lavin, M., Seibert, C. 2009. Grasses of Montana. Bozeman, MT: Montana State University Herbarium, gemini.oscs.montana.edu/~mlavin/herb/mtgrass.pdf.
- McGregor, R.L., and others. 1977. Atlas of the flora of the Great Plains. Ames, IA: Iowa State University Press.
- McGregor, R.L., Barkley, T.M. 1986. Flora of the Great Plains. Lawrence, KN: University Press of Kansas.
- Montana Natural Heritage program. 2002. List of Ecological Communities for Montana. Helena, MT: Montana Natural Heritage Program, Montana State Library (available at <http://nhp.nris.stat.mt.us>).
- Mueggler, W.F., Stewart, W.L. 1980. Grassland and shrubland habitat types of western Montana. USDA Forest Service General Technical Report INT-66. Ogden, UT: Intermountain Forest and Range Experimental Station.
- Pfister, R.D., Kovalchik, B.L., Arno, S.F., Presby, R.C. 1977. Forest habitat types of Montana. USDA Forest Service General Technical Report INT-34. Ogden, UT: Intermountain Forest and Range Experimental Station.
- Sims, P.L., Risser, P.G. 2000. Grasslands, pp. 324-356, *In* M.G. Barbour and W.D. Billings (eds.), North American Terrestrial Vegetation, second edition. Cambridge, U.K.: Cambridge University Press.
- University of Nebraska Press. 2005. The Journals of the Lewis and Clark Expedition. University of Nebraska-Lincoln Libraries-Electronic Text Center. <http://lewisandclarkjournals.unl.edu/>
- USDA, NRCS. 2008. The PLANTS Database (<http://plants.uds.gov>, 8 November 2008). Baton Rouge, LA: USDA National Plant Data Center.
- Van Bruggen, T. 1985. The vascular plants of South Dakota. Ames, IA: Iowa State University Press.
- Welsh, S.L., Atwood, N.D., Higgins, L.C., Goodrich, S. 2002. A Utah flora, 3rd ed. Provo, UT: Brigham Young University Print Services. 912 p.
- West, N.E., Young, J.A. 2000. Intermountain Valleys and Lower Mountain Slopes, pp. 256-284, *In* M.G. Barbour and W.D. Billings (eds.), North American Terrestrial Vegetation, second edition. Cambridge, U.K.: Cambridge University Press.

Long-term Browsing Impacts on Montana Ungulate Ranges

Scott K. Thompson¹ and Carl L. Wambolt²

ABSTRACT

*Ungulate impacts on woody vegetation have been a concern on Montana wildlife habitats for more than a half-century. Fenced areas restricting access of all ungulates (exclosures) were established between 1944 and 1988 on many habitats to evaluate ungulate impacts on shrubs. Our objective was to determine the effect of long-term browsing on big sagebrush (*Artemisia tridentata*), bitterbrush (*Purshia tridentata*), curlleaf mountain mahogany (*Cercocarpus ledifolius*) and horizontal juniper (*Juniperus horizontalis*). We hypothesized that the long-term effect of ungulate browsing would not impact these common shrubs. Canopy cover and density of shrubs were measured in and out of exclosures (n = 14) on environmentally paired sites. Big sagebrush canopy cover, density of mature big sagebrush, and production of winter forage (n = 7) were greater with protection on four sites ($P \leq 0.05$). Differences were not restricted to one subspecies of big sagebrush. Bitterbrush canopy cover and density of mature shrubs (n = 3) were greater with protection on two sites ($P \leq 0.05$). Curlleaf mountain mahogany canopy cover (n = 2) was greater with protection on both sites ($P \leq 0.05$), while density of mature mahogany was greater at one site ($P \leq 0.05$). Horizontal juniper cover (n = 2) was greater with protection at both sites ($P \leq 0.01$). We rejected our hypothesis at ten of the 14 sites evaluated. Long-term ungulate browsing has impacted shrubs at ten sites that are not geographically related. This has implications to plant communities and value of shrub habitats to wildlife.*

INTRODUCTION

Shrubs are a vital component of wildlife habitats by providing animals with necessary forage and cover. Shrubs due to their growth habits and nutritive properties provide valuable forage to browsing wildlife, and are an important food source during winter months (Bayless 1969; Cook 1972; Dorn 1970; Kasworm and others 1984; Kufeld 1973; Kufeld and others 1973; Lovass 1958; Martinka 1967; Ngugi and others 1992; Stevens 1970; Wilkins 1957; Urness 1989; Welch 1989). Shrubs often provide necessary thermal and security cover for wildlife according to growth habit and height (Carson and Peek 1987; Thomas and others 1979; Urness 1989). The sustainability of many wildlife species is dependent on the browse in their habitat. During the mid twentieth century, resource managers determined the need to study the impacts and extent of ungulate use of shrub communities. Most wild

ungulate populations across the western United States had been restored during the first half of the century. It became apparent that wild and domestic ungulate browsing, as well as drought, was altering shrub communities, therefore reducing their ability to sustain wildlife populations (Mussehl and Howell 1971). This was particularly important on wildlife winter ranges. To evaluate the influence of ungulates on shrubs and other plants, ungulate proof exclosures were constructed on many winter ranges and other areas of concern across Montana. These exclosures provide experimental comparisons of plant communities in the absence or presence of ungulate browsing (Daubenmire 1940; Young 1958). Our objective was to determine the impact of long-term browsing by ungulates on shrub species growing under a variety of environmental conditions and management scenarios. We examined big sagebrush (*Artemisia tridentata*), bitterbrush (*Purshia tridentata*), curlleaf mountain mahogany (*Cercocarpus ledifolius*) and horizontal juniper (*Juniperus horizontalis*) at 14 exclosure sites across Montana. Our objective was achieved by comparing shrub species protected inside exclosures to the same species browsed outside exclosures.

METHODS

Study Sites

Fourteen study sites were located throughout Montana on locations where shrub species of interest were present (table 1). Sites were chosen based on the presence of an exclosure functioning for a minimum of 10 years and the area providing important habitat to wild ungulates at least part of the year. The shrub species of interest at each site was the dominant shrub and had to be sufficiently dense and distributed both inside the exclosure and on the environmentally paired area outside the exclosure to warrant sampling. Confounding factors influencing only the browse community inside or only outside were taken into consideration when determining usability of a site.

Sampling

Comparisons were made by pairing the protected sites with environmentally similar unprotected sites. Considerations for pairing the sites environmentally were slope and aspect (Coughenour 1991). Pairing the sights by slope and aspect typically ensures similar soil types and microclimates. All measurements were taken at least 2 m from the exclosure fence to eliminate fence effects such as trailing. Sampling occurred from June through August in 2001 and 2002.

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Scott K. Thompson, Montana Fish Wildlife & Parks, Culbertson. sthompson@mt.gov

²Carl L. Wambolt, Montana State University, Bozeman. cwambolt@montana.edu

Canopy Cover

Shrub canopy cover was estimated using the line intercept method (Canfield 1941). Transect lines 30 m in length were established a minimum of 3 m apart. Transects were placed at regular intervals so that each transect represented an equal portion of the area compared (Coughenour 1991; Wambolt and Sherwood 1999). Exclosures were of varying size, so number of transects varied. Ten transects was the maximum required for an adequate sample on large sites (Wambolt and Sherwood 1999). As few as three transects may be adequate on smaller sites (Kay and Bartos 2000). Equal number of transects were established inside and outside each exclosure. Shrub canopy was measured in 3 cm increments along the line. A plumb bob was suspended from the line to accurately measure live shrub tissue intercepting the line. Shrub tissue intercepting less than 3 cm of the line was not included. Canopy cover of live shrub tissue for each species encountered was then calculated as a percent of the total line length. Lines were averaged together to obtain a mean canopy cover for shrubs inside or outside the exclosure.

Density

Density of shrubs was determined in 2 x 30 m belt transects. Line transects used for determining canopy cover provided the center of each belt. Belt transects were formed by measuring 1 m perpendicular to both sides of the line transect. The number of juvenile, mature, and dead shrubs rooted in the belt transect was recorded. Juvenile shrubs were those having an average crown cover < 15 cm from 4 axis measurements. Dead shrubs were those shrubs having no live tissue. Density for belt transects were then averaged together to obtain mean density for shrubs inside or outside the exclosure.

Sagebrush Winter Forage Production

Production of winter forage by big sagebrush taxa was estimated on seven sites using models developed by Wambolt and others (1994). Ten mature big sagebrush plants within each belt transect were measured for this parameter. Specific parameters measured on sagebrush plants varied by subspecies and form class (Wambolt and others 1994). Form classes depended upon historic low or high-use of big sagebrush. High use plants exhibit intense browsing by appearing "hedged". After obtaining density of mature big sagebrush and individual big sagebrush production of winter forage, the production of winter forage for big sagebrush per belt transect was calculated. Production inside exclosures was then compared to production outside on a kg/60m² basis.

Statistical Procedures

Rice and Westoby (1978) point out the difficulties of interpreting exclosures. Quadrants may be replicated inside

and outside exclosures, but not samples of the original vegetation subjected to an exclosure treatment. Location of an exclosure is typically arbitrary rather than random and management and grazing patterns vary between exclosures. Mackie (1973) cautioned against any generalizations concerning impacts of browsing due to variations in species, location, time, site or other environmental conditions affecting browse species productivity. For these reasons, sites were not combined, rather treated independently. We made no assumptions of pre-exclosure vegetation at the sites.

Canopy cover, density, and production of sagebrush winter forage inside the exclosure was compared to those parameters outside the exclosure at each site using a *t*-test. Equality of variances was tested using a Folded F test and variances were considered unequal when $P \leq 0.05$. A Student's *t*-test was used when variances were equal to determine differences between parameter means. When mean variances were unequal, the Satterthwaite *t*-value was used to determine differences between parameter means. Parameter means were considered significant when $P \leq 0.05$. Differences at the $P \leq 0.1$ level were also reported.

RESULTS AND DISCUSSION

Evaluation of shrub species at 14 study sites throughout Montana determined that ungulate browsing has impacted the dominant shrub species at ten sites (tables 2 and 3). Shrub parameters were greater ($P \leq 0.05$) with protection inside the exclosure than browsed outside exclosures in order for browsing to be considered to be an impact. Differences were detected for all shrub species. Exclosures where differences were detected were not restricted to specific regions of Montana or years the exclosure provided protection. Differences in browse parameters were found between inside and outside of the most recently constructed exclosure (1990) as well as the oldest exclosure (1945).

Differences in shrubs between excluded and protected areas developed from years of exclusion (tables 1 and 2). Intensity of ungulate browsing outside exclosures has likely been variable at some locations due to ungulate populations fluctuating through the years. A period of decreased browsing pressure may allow these communities to partially recover before the next period of intense browsing. Shrub communities outside exclosures have evolved in such a manner, while browsing inside exclosures is not a consideration in development of plants and their communities. The value in comparing protected to unprotected vegetation is to understand the site and plant community potential. In extreme cases, browse communities inside exclosures resulted in identification of a community that had been lost outside. Further browsing impacts may result in loss of shrubs in the plant community.

Table 1—Study site locations and descriptions.

Site	Environment Description	Years of Protection ^a	Latitude	Longitude	Elevation in Meters	Slope in Degrees	Aspect
1-Porcupine Creek	Mtn. Foothills	56	45.225805	-111.235500	2113	4	W-SW
2-Hyde Creek	Mtn. Foothills	39	45.016722	-111.716722	1972	8	E
3-Tepee Creek	Mtn. Foothills	46	45.070305	-111.164833	2143	16	S
4-Johnson Creek	Mtn. Foothills	35	45.835555	-113.022916	1880	10	S
5-Crooked Creek	Mtn. Foothills	40	45.093583	-108.414527	1930	8	E
6-Pine Ridge	Prairie Breaks	32	47.783777	-106.861805	740	5	SE
7-Cottonwood	Prairie Breaks	35	48.476638	-107.784027	704	3	SE
8-Poverty Flat	Montane Forest	43	48.919444	-115.206888	838	2	E
9-Young Creek	Montane Forest	43	48.991333	-115.193138	812	2	W
10-Dry Cottonwood	Mtn. Foothills	40	46.213555	-112.65675	1728	17	S
11-Scudder Creek	Mtn. Foothills	48	45.304805	-113.091888	2041	13	SE
12-Sykes Coulee	Mtn. Foothills	38	45.016611	-108.295138	1501	2	SE
13-Deer Hill	Mtn. Foothills	13	48.102111	-112.679305	1477	10	NW
14-Picket Pin	Mtn. Foothills	40	45.436361	-109.936027	1870	24	SW

^aNumbers of years site inside enclosure has been protected from browsing.

Big Sagebrush

Big sagebrush taxa were evaluated at seven sites. Protection from all ungulates from 32 to 56 years allowed big sagebrush to attain greater ($p \leq 0.01$) canopy cover at four sites (table 2; sites 1,3,5,7). Less significant ($p \leq 0.1$) differences in canopy were detected at a fifth site (site 4). Differences in canopy cover of big sagebrush between unprotected and protected areas were not restricted to one subspecies.

Rens (2001) reported canopy cover of mountain big sagebrush at five sites on the Northern Yellowstone Winter Range (NYWR) was greater ($P \leq 0.0001$) inside (20.0 percent) than outside (9.7 percent) enclosures. Wambolt and Sherwood (1999) also on the NYWR reported average canopy cover of big sagebrush at 19 sites was 19.7 percent inside big game enclosures and 6.5 percent outside. They reported areas protected from browsing had three times greater ($P \leq 0.0027$) canopy cover of big sagebrush than

unprotected areas. We found sites with canopy cover differences inside and outside enclosures averaged 15.5 percent and 6.9 percent, respectively and all big sagebrush sites averaged 12.3 percent inside and 7.1 percent outside.

Ungulate browsing also had an affect on density of big sagebrush plants. Four sites that had greater ($P \leq 0.01$) canopy cover of big sagebrush inside the enclosure also had greater ($P \leq 0.01$) density of mature big sagebrush inside the enclosure (table 2; sites 1,3,5,7). At site 4, density of mature big sagebrush inside the enclosure was greater ($P \leq 0.01$) than density outside, but canopy cover differences were less significant ($P \leq 0.1$). Our results agree with Wambolt and Sherwood (1999), as they reported a high association between big sagebrush cover and density ($r = 0.92$; $P \leq 0.000$). Density of mature Wyoming big sagebrush at site 6 was slightly higher ($P \leq 0.1$) outside the enclosure, which is attributed to competition with sweet clover inside the enclosure.

Table 2—Shrub canopy cover and density of mature shrubs at 14 environmentally paired sites either browsed or protected.

Site	Shrub Species	Percent Live Canopy Cover			Density of Mature ^a Shrubs/60m ²			Density of Young Shrubs/60m ²		
		Protected	↔ ^b	Browsed	Protected	↔ ^b	Browsed	Protected	↔ ^b	Browsed
1	Mt. Big Sagebrush	18.3	***	8.8	60.6	**	22.7	11.3		7.8
2	Mt. Big Sagebrush	9.5		8.7	24.4		25.7	4.3	**	13.3
3	Mt. Big Sagebrush	17.0	***	10.3	46.5	***	34.7	6.6		6.7
4	Mt. Big Sagebrush	11.1	*	8.4	50.4	***	36.7	6.5	***	17.3
5	WY Big Sagebrush	4.1	**	2.1	55.0	***	24.9	4.7		4.5
6	WY Big Sagebrush	3.9		5.3	15.6	*	33.2	2.1	**	5.5
7	WY Big Sagebrush	22.4	***	6.2	106.1	***	51.7	16.3		11.8
8	Bitterbrush	2.4	***	0.0	5.9	***	0.7	0.2		0.3
9	Bitterbrush	4.0		7.8	9.0	**	20.6	1.0		1.7
10	Bitterbrush	4.1	**	0.6	7.2	*	2.7	0.5		1.3
11	Curlleaf Mt Mahogany	34.9	***	11.0	31.3	**	14.1	2.0		0.1
12	Curlleaf Mt Mahogany	14.4	**	9.6	18.6		17.0	0.0		1.2
13	Horizontal Juniper	27.3	***	0.3						
14	Horizontal Juniper	47.1	***	6.0						

^aShrubs having a minimum canopy of 15 cm^bParamater comparison using *t*-test; **p* ≤ 0.1; ** *p* ≤ 0.05; *** *p* ≤ 0.01

Density of young big sagebrush was not greater ($P > 0.05$) with protection at any site (table 2). Snow may be protecting these young shrubs for several years, allowing them to grow to mature size before they are browsed (Mehus 1995; Wambolt and Sherwood 1999). Density of young big sagebrush was greater ($P \leq 0.05$) outside the enclosure at all sites where no differences in canopy cover were detected (table 2; sites 2,4,6). Protection afforded by snow and less browsing at these sites may allow more young shrubs to grow than at sites where ungulates are impacting sagebrush canopy cover.

Density of dead big sagebrush was unreliable as a comparison between protection and browsing. Some differences ($P \leq 0.01$) were detected at three sites (11, 16, 32) where big sagebrush canopy cover was also greater ($P \leq 0.01$) inside. Because cover and density are highly associated (Wambolt and Sherwood 1999), it may be logical that density of dead would also be higher. Additionally, dead shrubs may remain more recognizable and deteriorate slower in the absence of trampling by ungulates, thus estimation of their density may be higher inside exclosures.

Table 3—Production of big sagebrush winter forage at 7 environmentally paired sites either browsed or protected.

Site	Shrub Species	Kilograms of Winter Forage/60m ²		
		Protected	↔ ^a	Browsed
1	Mt. Big Sagebrush	3.7	**	1.3
2	Mt. Big Sagebrush	1.8		1.8
3	Mt. Big Sagebrush	3.6	***	1.8
4	Mt. Big Sagebrush	2.5		2.6
5	WY Big Sagebrush	0.6	***	0.2
6	WY Big Sagebrush	0.3		0.4
7	WY Big Sagebrush	1.9	***	0.6

^aParamater comparison using *t*-test; ** *p* ≤ 0.05, *** *p* ≤ 0.01

Estimates for winter forage production by big sagebrush were greater ($P \leq 0.05$) with protection at four of seven sites (table 3; sites 1,3,5,7). These sites also had greater percent canopy cover and density of mature big sagebrush inside the enclosure. Sites where differences were detected produced 60 percent more big sagebrush forage for winter per 60 m² inside than outside exclosures. Wambolt and Sherwood (1999) on the NYWR reported similar results. They reported protected mountain big sagebrush produced 88 percent more forage for winter than unprotected

sagebrush. Rens (2001) reported more than twice that (176 percent) when protected and unprotected mountain big sagebrush were compared.

Bitterbrush

Ungulate browsing has negatively impacted bitterbrush on two of the three sites evaluated (table 2; sites 8 and 10). At site 8, bitterbrush canopy cover and density of mature shrubs were greater ($P \leq 0.05$) inside the exclosure. Canopy cover was greater with protection at site 10, while differences in density were less significant ($P \leq 0.1$). At site 9, density was greater ($P \leq 0.05$) outside the exclosure, while there was no difference in canopy cover. Bitterbrush differences at site 9 may reflect apparent greater canopy cover and density of conifer trees inside the exclosure than outside.

Our results agree with other work done on bitterbrush including previous evaluation of sites 9 and 10 (Mackie 1973). Mackie (1973) also found greater density of bitterbrush outside at site 9 and greater inside at site 10. Unprotected areas had greater production of bitterbrush twigs than inside the exclosures at all 7 sites Mackie (1973) sampled. Mackie (1973) concluded differences in bitterbrush parameters were determined primarily by individual site characteristics and secondly by ungulate browsing. Kay (1995) found bitterbrush growing in the Greater Yellowstone Ecosystem, had less ($P \leq 0.05$) canopy cover, height, size, and volume outside a big game exclosure. He attributed this difference to repeated use by wild ungulates. The mixed results of our evaluation of bitterbrush also suggest other site factors in addition to browsing may explain some differences between protected and browsed sites.

Curleaf Mountain Mahogany

Canopy cover of curleaf mountain mahogany was greater ($P \leq 0.05$) inside the exclosure at both sites (11 and 12) where it was sampled. Differences in densities were not consistent between sites (table 2). Density of young mahogany was not different ($P \leq 0.05$) inside and outside exclosures at either site (table 2). Browsing has not limited recruitment of new mahogany plants. Density of mature and dead mahogany was greater ($P \leq 0.05$) inside the exclosure at site 11, but no differences were detected at site 12. Browsing appears to limit the size of individual shrubs at site 12, while canopy cover and number of mature shrubs were altered by ungulate browsing at site 11. Mackie (1973) studied site 11 and reported similar densities and canopy coverage but he emphasized that production of young twigs was greater outside the exclosure.

Horizontal Juniper

The two sites where horizontal juniper was evaluated had an exclosure protecting shrubs from all ungulate browsing and an exclosure only excluding livestock. Exclusion of all ungulates at site 13 for 13 years and at site 14 for 40 years

has allowed horizontal juniper to attain very high canopy cover (27.3 percent and 47.1 percent; table 2). Areas with exclusion of livestock only yielded canopy cover of 3.2 percent and 18.2 percent (figure 1). Areas accessible to all ungulates had canopy coverage of 0.3 percent and 6.0 percent (table 2). Differences between canopy cover of each phase of both exclosures were significant ($P \leq 0.05$; figure 1). These data suggest that the combined presence of wild ungulates and domestic ungulates has had the most detrimental effect on horizontal juniper, but wild ungulates alone have negatively impacted horizontal juniper.

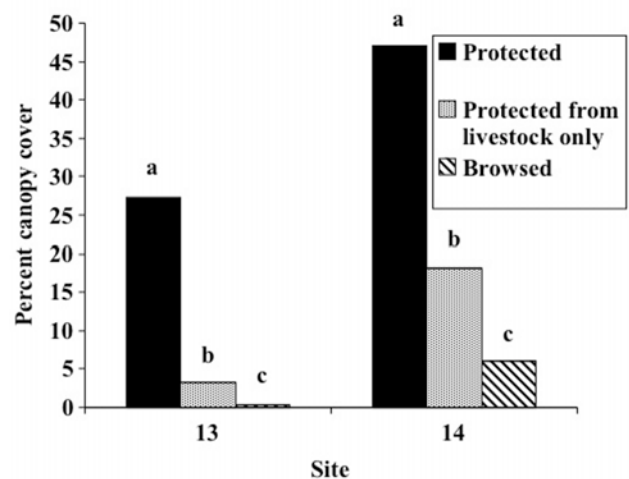


Figure 1—Percent canopy cover of horizontal juniper at 2 sites; environmentally paired either protected, protected from livestock or browsed. Sets of bars with different letter are significantly different ($p \leq 0.05$).

Kasworm and others (1984) studying ungulate diets near site 13, reported horizontal juniper as being very important forage to wintering elk (*Cervus elaphus nelsoni*), bighorn sheep (*Ovis canadensis*), and mule deer (*Odocoileus hemionus*). Presence of horizontal juniper in fecal samples of cattle (0.9 percent) suggests they consume only a small amount of it (Kasworm and others 1984). Cattle are much larger animals and consume greater quantities of forage than elk, bighorn sheep, and mule deer, so the actual amount of horizontal juniper consumed by cattle may be similar to that of wild ungulates. We also observed greater cover of grasses inside the livestock exclosures, which may conceal horizontal juniper directly or hold more snow during winter, thus preventing some use by wild ungulates. Livestock consume grass outside the exclosures, which may expose horizontal juniper and allow wild ungulates to browse juniper more readily. This theory contradicts Keigley and Olson (2001) where they observed horizontal juniper was more abundant inside a big game exclosure and outside, compared to inside the livestock exclosure. They suggest that a combination of reduced light availability

during the growing season due to tall grass, and browsing by wild ungulates in winter is restricting growth of horizontal juniper inside the livestock enclosure. Although our theory and Keigley and Olson's (2001) contradict each other, they agree with the findings of Kasworm and others (1984) that wild ungulates are heavily using horizontal juniper.

CONCLUSIONS

Concerns about browsing impacts have typically been centered on areas of special interest such as the Northern Yellowstone Winter Range (Kay 1995; Rens 2001; Ripple and Larsen 2000; Wambolt and Hoffman 2001; Wambolt and Sherwood 1999). Our study did not consider specific areas or taxa; rather, multiple shrub species across various environments with functional enclosures. Thus, this study shows the impacts of ungulates across many environmental and management scenarios.

Ten of the 14 study sites sampled provided evidence that long term browsing by ungulates is limiting the growth of dominant shrub. Logically, we would expect individual plants of landscape dominants to receive lighter levels of use, while plants of uncommon species might be more intensely used due to their rarity. We found this was not the case; rather browsing has impacted landscape dominants such as big sagebrush, bitterbrush, curlleaf mountain mahogany and horizontal juniper.

Wild and domestic ungulate populations on some of these areas should be re-evaluated. Impact on browse species has altered the vegetation of some ungulate ranges, which in turn may prove to be a long-term detriment to ungulate populations. Other wildlife species depending on these shrub communities have likely also been impacted. Consideration for shrub communities should be emphasized when managing ungulate populations.

REFERENCES

- Bayless, S.R. 1969. Winter food habits, range use, and home range of antelope in Montana. *Journal of Wildlife Management*. 33:538-551.
- Canfield, R.H. 1941. Application of the line interception method in sampling range vegetation. *Journal of Forestry*. 39:388-394.
- Carson, R.G., Peek, J.M. 1987. Mule deer habitat selection patterns in north central Washington. *Journal of Wildlife Management*. 51:46-51.
- Cook, W.C. 1972. Comparative nutritive values of forbs, grasses and shrubs, p. 303-310. *In*: McKell, C.M.; Blaisdell, J.P.; Goodin, J.R., editors. *Wildland shrubs – their biology and utilization*. USDA, Forest Service General Technical Report. INT-1. Ogden, UT.
- Coughenour, M.B. 1991. Biomass and nitrogen responses to grazing of upland steppe on Yellowstone's Northern Winter Range. *Journal of Applied Ecology*. 28:71-82.
- Daubenmire, R.F. 1940. Enclosure technique in ecology. *Ecology*. 21:514-515.
- Dorn, R.D. 1970. Moose and cattle food habits in southwest Montana. *Journal of Wildlife Management*. 34:559-564.
- Kasworm, W.F., Irby, L.R., Ihlsle Pac, H.G. 1984. Diets of ungulates using winter Ranges in North central Montana. *Journal of Range Management*. 37:67-71.
- Kay, C.E. 1995. Browsing by native ungulates: Effects on shrub and seed production in the Greater Yellowstone Ecosystem, p. 310-320. *In*: Roundy, B.A., McArthur, E.D., Haley, J.S., Mann, D.K., compilers. 1995. *Proceedings: Wildland shrub and arid land restoration symposium*; 1993 October 19-21; Las Vegas, NV. Gen. Tech. Rep. INT-GTR-315.
- Ogden, UT: Forest Service, Intermountain Research Station. 384 p.
- Kay, C.E., Bartos, D.L. 2000. Ungulate herbivory on Utah aspen: Assessment of long-term enclosures. *Journal of Range Management*. 53:145-153.
- Keigley, R.B., Olson, G.R. 2001. The condition of browse plants at the Theodore Roosevelt Memorial Ranch (TRMR), p. 65-70. *In*: Knapp, S.J., Frisina, M.R., editors. *Statewide browse evaluation*. Montana Fish Wildlife & Parks, Project Rep. No. 1.
- Kufeld, R.C. 1973. Foods Eaten by the Rocky Mountain Elk. *Journal of Range Management*. 26:106-113.
- Kufeld, R.C., Wallmo, O.C., Feddema, C. 1973. Foods of the Rocky Mountain Mule Deer. USDA, Forest Service, Research Paper RM-111.
- Lovass, A.L. 1958. Mule deer food habits and range use, Little Belt Mountains, Montana. *Journal of Wildlife Management*. 22:275-282.
- Mackie, R.J. 1973. Evaluation of range survey methods, concepts and criteria (Responses of four browse species to protection on big game winter ranges in western Montana). Unpublished report. 1973. Montana Fish Wildlife & Parks Project. W-120-R-3. Helena MT.
- Martinka, C.J. 1967. Mortality of northern Montana pronghorns in a severe winter. *Journal of Wildlife Management*. 31:159-164.
- Mehus, C.A. 1995. Influences of browsing and fire on sagebrush taxa of the Northern Yellowstone Winter Range. M.S. Thesis. Montana State University, Bozeman, MT.
- Musshel, T.W., Howell, F.W. 1971. Game Management in Montana. Montana Fish Wildlife & Parks Federal Aid Project W-3-C. Helena, MT.
- Ngugi, K.R., Powell, J., Hinds, F.C., Olson, R.A. 1992. Range animal diet composition in south central Wyoming. *Journal of Range Management*. 45:542-545.
- Rens, R.J. 2001. Elk effects on sagebrush-grassland after fire on Yellowstone's Northern Range. M.S. Thesis. Montana State University, Bozeman, MT.
- Rice, B., Westoby, M. 1978. Vegetative responses of some Great Basin shrub communities protected against jackrabbits or domestic stock. *Journal of Range Management*. 31:24-28.
- Ripple, W.J., Larsen, E.J. 2000. Historic aspen recruitment, elk, and wolves in northern Yellowstone National Park, USA. *Biological Conservation*. 95:361-370.
- Stevens, D.R. 1970. Winter ecology of moose in the Gallatin Mountains, Montana. *Journal of Wildlife Management*. 34:37-46.
- Thomas, J.W., Black, H. Jr., Sherzinger, R.J., Pedersen, R.J. 1979. Deer and elk, p. 104-127. *In*: Thomas, J.W., editor. *Wildlife habitats in managed forest: the Blue Mountains of Oregon and Washington*. Forest Service Agriculture Handbook. No. 553.
- Urness, P.J. 1989. Shrubs as habitats for wildlife, p. 441-458. *In*: McKell, C.M., editor. *The biology and utilization of shrubs*. Academic Press Inc., San Diego. CA.
- Wambolt, C.L., Sherwood, H.S. 1999. Sagebrush response to ungulate browsing in Yellowstone. *Journal of Range Management*. 52:363-369.
- Wambolt, C.L., Hoffman, T.L. 2001. Mountain Big Sagebrush Age Distribution and Relationships on the Northern Yellowstone Winter Range. *In*: McArthur, E.D., Fairbanks, D.J., compilers. 2001. *Proceedings: Shrubland ecosystem genetics and biodiversity symposium*; 2000 June 13-15; Provo, UT. RMRS-P-21. Ogden, UT: USDA, Forest Service, Rocky Mountain Research Station.
- Wambolt, C.L., Creamer, W.H., Rossi, R.J. 1994. Predicting big sagebrush winter forage by subspecies and browse form class. *Journal of Range Management*. 47:231-234.
- Welch, B.L. 1989. Nutritive value of shrubs, p. 405-424. *In*: McKell, C.M., editor. *The biology and utilization of shrubs*. Academic Press Inc., San Diego. CA.
- Wilkins, B.T. 1957. Range use, food habits, and agricultural relationships of the mule deer, Bridger Mountains, Montana. *Journal of Wildlife Management*. 21:159-168.
- Young, S. 1958. Enclosures in big game management in Utah. *Journal of Range Management*. 11:186-190.

Sage-grouse (*Centrocercus urophasianus*) Habitat in Central Montana

Jennifer Woodward¹, Carl Wambolt², Jay Newell³, and Bok Sowell⁴

ABSTRACT

Greater sage-grouse (*Centrocercus urophasianus*) habitat was studied in central Montana primarily on Wyoming big sagebrush (*Artemisia tridentata* Nutt. ssp. *wyomingensis* Beetle & Young) dominated rangeland. The primary objective was to compare shrub and herbaceous parameters within (use, random or non-use) and between seasonal habitats (nest, brood, winter). Two study sites (Musselshell and Golden Valley counties), and 2 years (2004 and 2005) were compared. Nest, brood, and random sites were compared for herbaceous cover, and grass height. Nest, brood, random, winter use and winter non-use sites were evaluated for shrub cover, density, and height. All differences were considered significant at $P \leq 0.05$. Sage-grouse nested in areas with greater total shrub cover and height, and taller live and residual grass than was randomly available. No differences were found between brood and paired random sites for any of the herbaceous or shrub parameters measured. Shrub cover and density were greater at winter use sites than non-use sites. Winter use sites had less shrub cover than nest sites. The nest and brood habitat had similar shrub cover, density, and height on the study area. Sage-grouse habitats should be managed to include sagebrush, forbs, and grass. Herbaceous vegetation was more important during nesting and brood rearing than during the winter. Therefore, some portions of grouse habitat may benefit from management for greater herbaceous cover, but never at the sake of less sagebrush. Sagebrush cover from 10 to 30 percent was the most consistent component of sage-grouse habitat.

INTRODUCTION

It is estimated that the overall distribution of sage-grouse (*Centrocercus* spp.) has decreased by 50 percent since European settlement (Braun 1998). The historical distribution of greater sage-grouse (*C. urophasianus*) included 12 states and 3 Canadian provinces; currently sage-grouse have been extirpated from British Columbia and Nebraska (Schroeder and others 2006).

Sage-grouse decline is most often attributed to a loss of habitat. Sage-grouse are sagebrush obligates, dependent on sagebrush for food, thermal cover, and hiding cover (Patterson 1952; Wallestad 1975). Montana currently supports 11 million ha of sage-grouse habitat (Montana

Sage-grouse Work Group 2002); although it is estimated 50 percent of the original habitat has been lost (C.L.Wambolt personal communication, March 2005).

Sage-grouse habitat is quite variable due to variations in weather (especially snow depth), sagebrush taxa present, habitat types and patterns available, and topography of habitat (Eng and Schladweiler 1972; Wallestad 1975; Remington and Braun 1985; Connelly and others 2000). Few studies have had the opportunity to concurrently investigate nesting, brood rearing and winter habitats required by sage-grouse. Further information on sage-grouse in Montana would assist land managers in making decisions in favor of sage-grouse.

The primary objective was to compare shrub and herbaceous parameters within (use, random or non-use) and between seasonal habitats (nest, brood, winter) and 2 years (2004 and 2005).

METHODS

Study Area Description

This study was conducted in central Montana with one location in Musselshell county and another in Golden Valley county, 25 km north and 32 km west of Roundup. The majority of the study area is privately owned (75 percent), with some land managed by the Bureau of Land Management (16 percent), and the remaining by the state (5 percent). Eighty-nine percent of the study area is used for range, beef and sheep production. The weather station in Roundup receives a mean annual precipitation of 32 cm (Western Regional Climate Center 2005). During the growing season (April to September) the area receives 75 percent of the annual mean average precipitation. The study area is made up of shale lowlands separated by sandstone ridges containing several undrained basins (USDA 2003). The study sites consisted of the Wyoming big sagebrush (*Artemisia tridentata* Nutt ssp. *wyomingensis* Beetle & Young) habitat type, greasewood (*Sarcobatus vermiculatus* (Hook.) Torr.) bottoms, native and introduced grasslands, and agricultural lands. This study was conducted in conjunction with Sika (2006) whose study focused on breeding ecology, survival rates and mortality causes of sage-grouse on the same study area.

Collection of Bird Locations

Over a 3 year period (2003 to 2005) sage-grouse were captured on or near leks during breeding season and fitted with a radio transmitter. A receiver was used to precisely locate sage-grouse and a geographic position system (GPS)

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Woodward, Jennifer Natural Resource Conservation Service, White Sulphur Springs MT

²Carl Wambolt, Animal and Range Sciences Department, Montana State University, Bozeman, MT

³Jay Newell, Montana Fish Wildlife and Parks, Roundup MT

⁴Bok Sowell Animal and Range Sciences Department, Montana State University, Bozeman, MT

was used to record exact location of tracks, fresh sage-grouse droppings, or nest location. Use sites were relocated with GPS in order to conduct habitat measurements. Winter sites were located 3 to 6 months, and nest and brood sites within 7 days after bird location was collected via GPS.

MEASUREMENTS

Nesting Habitat

Sagebrush canopy cover, density, and height, forb, grass, and herbaceous cover, and live and residual grass height were measured at each nest site. The height and average diameter of the shrub the hen nested under was determined. Nest shrub productivity was calculated by measuring the nest shrub's major axis, followed by a perpendicular minor axis, and 2–45° crown width measurements, and is reported in grams of available winter forage (Wambolt and others 1994). All measurements were conducted on 2 perpendicular 30 m N-S and E-W transects, with the nest located at the center (15 m point) of each line.

Line intercept sampling was used to estimate shrub canopy cover, and height along transects (Canfield 1941, Connelly 2003). Gaps in shrubs >3 cm (0.1 ft) were excluded. Shrub height was measured every 2 m on the nearest shrub with an average crown >15 cm in diameter ($n = 15/\text{transect}$). Belt transects were used to estimate shrub density (Connelly and others 2003). All shrubs >15 cm in diameter and rooted within 1 m of either side of the 3 to 30 m transect lines were recorded. Total herbaceous forb and grass cover were measured using 20 x 50 cm quadrats (Daubenmire 1959). A total of 20 quadrats were measured along each of the two transects. Vegetative droop height of living grass and residual grass was also recorded at each quadrat.

Random sites were paired with nest sites within the same habitat to test if hens were randomly selecting shrub or herbaceous characteristics for nesting. Shrub and herbaceous measurements were completed using the same methods as nest sites. Random sites were measured on the same or next day as their paired nest sites. At each nest site a random compass direction and distance (between 30 m and 1000 m) were chosen using random number tables. The tallest sagebrush, nearest the end of the random distance was selected as the random nest shrub. If the habitat encountered at the random site was not sagebrush, such as an agricultural field, the closest sagebrush stand in the same direction was selected.

Brood Rearing Habitat

Hens with broods were tracked throughout the brood-rearing season. Shrub and herbaceous parameters at brood sites were measured using the same methods as nest sites. Paired random locations were selected in the same way as

nest sites. Brood locations were located and measured at 1 and 4 weeks after hatch.

Winter Habitat

Winter habitat measurements were collected along 3 to 30 m transects. The sampling layout differed from nest and brood sites because typically during the winter birds are in flocks that are spread out across the landscape. The first transect was centered over droppings found at the GPS location of the use site. The second and third transects were 30 m to each side and parallel to the first transect. Shrub measurements including cover, density and height were completed precisely the same as for nest sites. Residual grass height (including inflorescence) and cover were measured at 2005 to 2006 winter use and non-use sites.

Sage-grouse winter non-use sites were established during the summer of 2005 and the winter of 2005 to 2006. Random points were placed on ArcGIS 9[®] within areas that were not known to be used heavily during winter. Random points were only placed in soil types that grouse had used during our study. No random points were placed within 300 m of the actual winter bird location.

A GPS unit was used to locate sampling sites. An area of 1200m² was delineated around each GPS location for sampling. This area was searched thoroughly for droppings; if droppings were found the site was dropped from our analysis. Droppings were persistent for at least 6 months. At locations with no indication of use, 3 30 m transects, duplicating those in use sites were sampled. Sites with < 2 percent sagebrush were discarded and not sampled as non-use sites.

ANALYSES

An analysis of variance (ANOVA) was conducted using the GLM version of SAS[®] 9.0 to determine if sage-grouse habitat differed by season (nest, brood, random, winter, or winter non-use), or year (2004 and 2005). Nest, brood, and random sites were compared for total herbaceous, grass, and forb cover, and live and residual grass height. Nest and paired random sites were evaluated for nest shrub height, nest shrub productivity and crown diameter of nest shrub. Nest, brood, random, winter use and winter non-use sites were compared for total shrub canopy cover, total shrub density, and shrub height. The experimental unit was each bird location, or one sampled site, all transects from each site were averaged together. Effects in the model were compared with least squares means. Because data were unbalanced a Tukey-Kramer test was used to separate least squares means (Kramer 1959). The linear model included all main effects (season, year, county) and all possible interactions. When interaction terms were not significant reduced models were fitted to the data. Least squares means are presented where no interactions were determined. Least

squares means for the interactive terms are reported when interactions were significant. Residual grass height and cover from the 2005 to 2006 winter were analyzed with either 2-independent sample t-tests or Wilcoxon-Mann-Whitney tests depending on normality. These data could not be added to the ANOVA because of differences in sampling methodology. All differences were considered significant at $P \leq 0.05$.

Vegetative parameters were analyzed for 648 sites measured in 2004 and 2005. Sites with 0 percent shrub cover were omitted from the analyses. Included in the analysis were 22 sites where greasewood contributed >50 percent of total canopy cover, and 4 sites where silver sage contributed >50 percent total canopy cover. The remaining 626 sites (97 percent) sampled had >50 percent shrub cover of Wyoming big sagebrush. Wyoming big sagebrush, silver sagebrush, and greasewood sites were analyzed together as one shrub cover type in the ANOVA. Over 90 percent of shrub cover was from Wyoming big sagebrush.

RESULTS AND DISCUSSION

Nesting Habitat

The importance of herbaceous and shrub cover for nest concealment has been demonstrated in previous studies (Connelly and others 2000, Sveum and others 1998). Sage-grouse nested in areas with greater total shrub cover (15 percent vs. 13 percent; $P < 0.03$) and height (28 cm vs. 26 cm; $P < 0.05$), and taller live (12 cm vs. 11 cm) and residual grass (9 cm vs. 8 cm) than was randomly available in our study. When total shrub cover, residual grass height, or residual grass cover increased by 10 percent, the probability of a nest increased by 31.3 percent (Holloran et. al. 2005). The shrub under which the hen nested was taller (50 cm vs. 44 cm; $P < 0.001$) and had greater productivity (61 g vs. 51 g; $P < 0.001$) than random shrubs in both years.

Due to increased precipitation in 2005, total herbaceous (18 percent vs. 13 percent), grass (15 percent vs. 12 percent), and forb cover (7 percent vs. 3 percent), and live grass height (13 cm vs. 10 cm) were greater ($P < 0.001$) in 2005 than 2004. Average crown diameter was greater at nest sites than random sites in 2004 but similar in 2005 (figure 1). Sika (2006) found weak support for increased nest success in 2005 on the same study area. Coggins (1998) found nest success for sage-grouse increased in years with greater residual tall grass cover. Haustliner (2003) considered grass height to be a limiting factor for nest success during drought years. Herbaceous cover associated with nest sites may have provided scent, visual, and physical barriers to predators (DeLong and others 1995).

There was a greater number of re-nests (ie. 2nd and 3rd nest attempts) in 2005 than 2004. Re-nests were more successful (56 percent) than first nests (32 percent) (Sika 2006). This

was likely because of the increase in herbaceous cover. Although there was a higher proportion of successful nests in 2004, there was greater hen success in 2005 (success in raising a brood out of all of the females beginning the study) (Sika 2006). Moynahan (2004) found that nesting probability increased when range condition was improved.

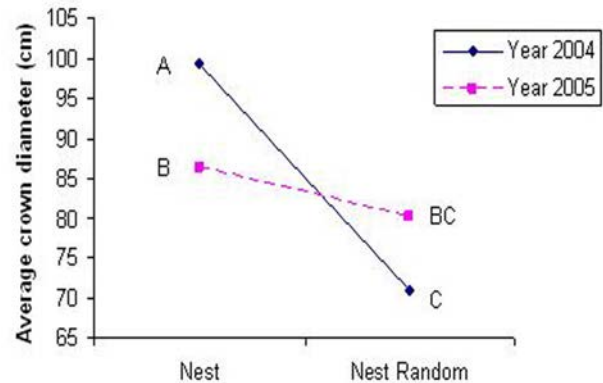


Figure 1—Interaction between year and seasonal habitat for average crown diameter of the nest shrub for nest and paired random sites in 2004 and 2005 in Golden Valley and Musselshell counties. Means differ ($P \leq 0.05$) between sites when followed by a different letter.

Brood Rearing Habitat

No differences were found between brood and paired random sites for any of the herbaceous or shrub parameters measured on our study. Klebenow (1969) did not find a statistical difference between brood and random sites in Idaho. A meta-analysis of 7 sage-grouse studies throughout the west found areas used by broods < 6 weeks of age had greater forb and grass cover, taller grass, and less sagebrush cover than random sites (Hagen and others 2007).

There was greater shrub height (29 cm vs. 25 cm), total herbaceous cover (19 percent vs. 16 percent), forb cover (15 percent vs. 13 percent), and live grass height (17 cm vs. 11 cm) in 2005 than 2004 ($P < 0.001$). Shrub density ($1.5/\text{m}^2$ vs. $1.1/\text{m}^2$) and residual grass height (9 cm vs. 5 cm) were greater in 2004 than 2005, indicating broods used different areas in years with different precipitation. Broods in southern Oregon and southern Idaho used areas with the greatest forb cover (Klebenow 1969, Drut 1994a). Forbs, insects, and sagebrush are the primary food sources for young chicks (Drut 1994b, Peterson 1970). Forb cover increased 2.5 fold from 2004 to 2005 due to increased precipitation across my study area. As forb cover increased, broods may have been able to forage in a variety of cover types in 2005. The added herbaceous cover may have added a component making chicks less visible in a variety of cover types. Coggins' (1998) brood use was equitably distributed between cover types in wet years more than dry years because of the greater availability of forbs.

Table 1— Maximum, minimum, and average temperature, total precipitation, and total snow depth from January to December in 2004 to 2006 at station 247220 in Roundup, MT (National Oceanic and Atmospheric Administration 2006).

Year	Month	Average Maximum Temp °F	Average Minimum Temp °F	Average Temp °F	Total Precip (in)	Total Snow (in)
2004	Jan	33	12	22.5	0.40	7.5
2004	Feb	43	19	31	0.26	0.5
2004	Mar	60	30	45	0.11	0.5
2004	Dec	46	21	33.5	0.39	4.2
2005	Jan	33	11	22	0.26	6.5
2005	Feb	49	19	34	Trace	Trace
2005	Mar	54	27	40.5	0.63	6.4
2005	Dec	36	18	27	0.33	4.5
2006	Jan	47	28	37.5	0.02	0.0
2006	Feb	42	16	29	0.22	1.4
2006	Mar	47	22	34.5	1.55	8.0

Increased food availability and cover have the potential to increase survival of chicks (Crawford 1998). On our study area there was 71 percent brood survival in 2004, and 84 percent in 2005 (brood survival is the survival of at least one chick to 30 days of age) (Sika 2006). Increased grass height, total herbaceous cover and forb cover, likely increased chick survival.

Due to unfavorable precipitation and a lack of forbs, the food source for chicks could have been comprised more of sagebrush than forbs in 2004. This could be why sagebrush density was higher in 2004 than 2005. Drut (1994a) found chick diets in Oregon to correspond directly to the availability of primary foods. Forbs and invertebrates constituted >75 percent of the diet at Hart Mountain, while sagebrush composed 65 percent of the mass consumed by chicks at Jackass Creek. Peterson (1970) in central Montana and Klebenow (1968) in Idaho found young grouse ate mostly forbs and insects and very little sagebrush until they were ≥ 10 weeks old.

Winter Habitat

Both winters were relatively mild with no snow accumulations >10 cm on days we collected bird locations and no more than 20 cm of snow fell within a month (table 1). Shrub height was greater at winter non-use sites than use sites in 2005 (36 cm vs. 32 cm; $P < 0.01$), but similar in 2004 (27 cm vs. 27 cm). Most winter studies have found height of sagebrush to be an important determinant of sage-grouse habitat because sagebrush availability above the snow pack has been determined to be critical for sage-grouse survival (Wallestad 1975, Beck 1977, Robertson 1991). Snow never completely covered sagebrush during the 2 years of this study. In a study in the Yellow-Water Triangle of Montana, <15 miles north of our Musselshell study site, Wallestad (1975) found sage-grouse restricted to 7 percent of their available range when snow depths exceeded 30 cm. This

occurred in 3 of the 7 winters of his study. It is not known where grouse will go under harsh winter conditions on our study area.

Despite our sampling occurring during mild winters, shrub cover (12 percent vs. 10 percent) and density ($1.2/\text{m}^2$ vs. $0.8/\text{m}^2$) were greater ($P < 0.001$) at winter use sites than non-use sites. This is similar to findings in Idaho sage-grouse winter habitats (Robertson 1991). During the winter sage-grouse were almost completely reliant on sagebrush as a food and cover source (Patterson 1952, Wallestad 1975). Sage-grouse were able to maintain or gain weight during the winter (Patterson 1952, Remington and Braun 1988) because of their nutritious diet (Wambolt 2004) of abundant sagebrush.

Shrub density and cover were greater ($P < 0.001$) in 2004 than 2005. Winter weather conditions were similar for both years. In the fall of 2005, 800 ac of important winter habitat in Musselshell County were plowed, making it unavailable for use during the 2005 to 2006 winter. While it appeared that birds had plenty of similar habitats available they were still found in the plowed area and in adjacent habitat that appeared less than ideal. The 10 sites immediately surrounding the newly plowed area during the 2005 winter had a shrub canopy cover of 7.2 percent and a density of 0.86 shrubs/ m^2 . I found sage-grouse returned to wintering grounds year after year as did Berry and Eng (1985). Swenson (1987) in south-central Montana found that grouse populations declined by 73 percent after 30 percent of a known sage-grouse winter habitat was plowed. On our study area, lek numbers increased the year following the plowing (MTFWP unpublished data), possibly due to the wet productive spring before. Declines in population numbers could occur in the future if the study area received harsh winter conditions.

Sika (2006) found monthly winter survival to be lower in the 2005 winter than the 2004 winter on our study area. Because reproductive effort was higher in 2005, Sika (2006) attributed the decline in survival to trade-offs between survival and reproductive effort. The differences in habitat between the 2 years (decreased density and cover in 2005) could have been related to decreased survival in 2005. It is possible that grouse choose habitats that were less suitable causing mortality. It is also possible that grouse choose habitats to mitigate high mortality, for example they may have used habitats with less density and cover in order to be able to visualize and avoid predators.

Residual grass height (19 cm vs. 18 cm) and cover (13 percent vs. 14 percent) were similar at winter use and non-use sites. Because grass heights measured for residual grass height in the winter of 2005 to 2006 included inflorescences the heights are not comparable with nest and brood residual grass heights were vegetative droop heights were measured. Sage-grouse did not appear to be selecting areas with vegetative cover in the winter. To our knowledge, no other study has addressed the importance of herbaceous cover in the winter. Selection for areas with greater herbaceous cover seems to be restricted to nesting and brood rearing activities when predator avoidance requires hiding.

Our winter use sites had less shrub cover ($P<0.001$) than nest sites (12 percent vs. 15 percent), seemingly in conflict with Wallestad (1975) who found nest and winter habitats to be synonymous in central Montana. We believe our winter sites were low in cover, height, and density compared to other studies because snow depth was not great enough during this study to influence grouse distribution. Wallestad's (1975) study included 3 of 7 winters which were above average in snow accumulation. If our study had occurred in similar winters it is likely that the winter habitat could have been similar to nesting habitat.

Other studies have found brood sites to have less shrub cover than random sites and nest sites to have greater shrub cover than random sites (Hagen and others 2007); however we found the 2 habitats (nest and brood) to be similar in our study area. We removed all non-shrub habitats from our analysis in order to compare shrub habitats; if this had not been done the average cover of shrubs at brood sites would have been lower. As forbs desiccate, grouse often moved to more mesic sites in other studies (Connelly and others 2000). Birds in our study were non-migratory with movements typically less than 10 km between seasonal habitats, therefore, nesting and brood rearing routinely took place within close proximity.

Logically, because of difference in season, brood sites had greater grass height than nest sites and nest sites had greater residual grass height than brood sites ($P<0.02$). Total herbaceous cover and forb cover were greater ($P<0.04$) at brood sites, but grass cover was similar at nest and brood sites. Residual grass height was lower at brood sites

because live grass was providing greater cover than residual grass and because residual grass was continually degrading as the season progressed.

Wallestad (1975) analyzed sage-grouse production over 10 years and found that rain during the laying season resulted in poor production due to a late hatch, but spring rainfall increased production overall. The spring of 2005 was rainy during the laying period in early May and until the end of June. More nests failed in 2005 than 2004, but the hens were able to re-nest, and re-nests were more successful (Sika 2006). The continuing rain may have enabled the green up period to last long enough to promote chick survival. Due to increased precipitation, total herbaceous, grass and forb cover were greater in 2005 than 2004 ($P<0.05$). The proportion of hens that successfully raised a brood to 30 days of age increased from 28 percent in 2004 to 43 percent in 2005 (Sika 2006). Lek counts increased between 30 to 50 percent from 2005 to 2006 (MTFWP unpublished data) verifying high recruitment from the 2005 breeding season. Moynahan (2004) also observed greater re-nesting in a year when 1st nest survival was low to moderate and habitat conditions were favorable.

CONCLUSION

Sage-grouse habitats should be managed to include sagebrush, forbs, and grass. Herbaceous vegetation was more important during nesting and brood rearing than during the winter. Therefore, some portions of grouse habitat may benefit from management for greater herbaceous cover, but never at the expense of sagebrush. Sagebrush cover from 10 to 30 percent was the most consistent component of sage-grouse habitat.

REFERENCES

- Beck, T.D.I. 1977. Sage grouse flock characteristics and habitat selection during winter. *Journal of Wildlife Management* 41:18-26.
- Berry, J.D., Eng, R.L. 1985. Interseasonal movements and fidelity to seasonal use areas by female sage grouse. *Journal of Wildlife Management* 49:237-240.
- Canfield, R.H. 1941. Application of the line interception method in sampling range vegetation. *Journal of Forestry* 39: 388-394.
- Connelly, J.W. et al. 2000. Guidelines to manage sage grouse populations and their habitats. *Wildlife Society Bulletin* 28: 967-985.
- Connelly, J.W., Reese, K.P., Schroeder, M.A. 2003. Monitoring of greater sage-grouse habitat and populations. University of Idaho, Experimental Station Bulletin 80, Moscow. 50p.
- Coggins, K.A. 1998. Relationship between habitat changes and productivity of sage grouse at Hart Mountain National Antelope Refuge, Oregon [thesis]. Corvallis, OR: Oregon State University. 85 p.
- Crawford, J.A. et al. 2004. Ecology and management of sage-grouse and sage-grouse habitat. *Journal of Range Management* 57:2-19.
- Daubenmire, R.F. 1959. A canopy-coverage method of vegetation analysis. *Northwest Science* 33: 43-64.
- DeLong, A.K., Crawford, J.A., DeLong, D.C. 1995. Relationships between vegetational structure and predation of artificial Sage Grouse nests. *Journal of Wildlife Management* 59:88-92.

- Drut, M.S., Crawford, J.A., Gregg, M.A. 1994a. Brood habitat use by sage grouse in Oregon. *Great Basin Naturalist* 54:170-176.
- Drut, M.S., Pyle, W.H., Crawford, J.A. 1994b. Technical note: Diets and food selection of sage grouse chicks in Oregon. *Journal of Range Management* 47:90-93.
- Eng, R.L.P., Schladweiler, 1972. Sage grouse winter movements and habitat use in central Montana. *Journal of Wildlife Management*. 36: 141-146.
- Hagen, C.A., Connelly, J.W., Schroeder, M.A. 2007. A meta-analysis of greater sage-grouse *Centrocercus urophasianus* nesting and brood rearing habitats. *Wildlife Biology*. 13(Suppl. 1):42-50.
- Hausleitner, D. 2003. Population dynamics, habitat use and movements of greater sage-grouse in Moffat County, Colorado [thesis]. Moscow, ID: University of Idaho.
- Holloran, M.J., Heath B.J. 2005. Greater sage-grouse nesting habitat selection and success in Wyoming. *Journal of Wildlife Management* 69:638-649.
- Kramer, C.K. 1956. Extension of Multiple range tests to group means with unequal numbers of replication. *Biometrics* 12:307-310.
- Klebenow, D.A., Gray, G.M. 1968. Food habits of juvenile Sage Grouse. *Journal of Range Management* 21:80-83.
- Klebenow, D.A. 1969. Sage Grouse nesting and brood habitat in Idaho. *Journal of Wildlife Management* 33:649-662.
- Moynahan, B.J. 2004. Landscape-scale factors affecting population dynamics of greater sage grouse (*Centrocercus urophasianus*) in north-central Montana [dissertation] Missoula, MT: University of Montana.
- National Oceanic Atmospheric Administration. 2006. Annual climatological summary, Roundup, Montana. Available at: <http://cod.ncdc.noaa.gov/ancsum/ACS>. Accessed August 3rd, 2006.
- Patterson, R.L. 1952. The sage grouse in Wyoming, Sage Books, Inc., Denver Colo.
- Peterson, J.G. 1970. The food habits and summer distribution of juvenile sage grouse in central Montana. *Journal of Wildlife Management* 34: 147-155.
- Remington, T.E., Braun, C.E. 1985. Sage grouse food selection in winter, North Park Colorado. *Journal Wildlife Management* 49:1055-1061.
- Remington, T.E., Braun, C.E. 1988. Carcass composition and energy reserves of sage grouse during winter. *Condor* 90:15-19.
- Robertson, M.D. 1991. Winter Ecology of migratory sage-grouse and associated of prescribed fire in south eastern Idaho [thesis]. Moscow, ID: University of Idaho.
- Sika, J.L. 2006. Breeding ecology, survival rates and mortality causes of hunted and non-hunted greater sage-grouse in central Montana [thesis]. Bozeman, MT: Montana State University. 97p.
- Schroeder, M.A., Connelly, J.W., Wambolt, C. L., Braun, C.E., Hagen, C.A., Frisina M.R. 2006. Society for Range Management Issue Paper: Ecology and Management of sage-grouse and sage-grouse habitat- a reply. *Rangelands* 28(3) 3-7.
- Sveum, C.M., Edge, W.D., Crawford, J.A. 1998. Nesting habitat selection by Sage Grouse in south-central Washington. *Journal of Range Management* 51:265-269.
- Swenson, J.E., Simmons, C.A., Eustace, C.D. 1987. Decrease of sage grouse *Centrocercus urophasianus* after ploughing of sagebrush steppe. *Biological Conservation*. 41:125-132.
- USDA. 2003. Soil survey of Musselshell County, Montana. Natural Resource Conservation Service office, Bozeman, MT. 217 p.
- Wallestad, R.O. 1975. Life history and habitat requirements of sage-grouse in central Montana. Montana Fish and Game Department, Technical Bulletin, Helena, USA.
- Wambolt C.L. 2004. Browsing and plant age relationships to winter protein and fiber of big sagebrush subspecies. *Journal of Range Management* 57:620-623.
- Wambolt, C.L., Creamer, W.H., Rossi, R.J. 1994. Predicting big sagebrush winter forage by subspecies and browse form class. *Journal of Range Management* 47:231-234.
- Western Regional Climate Center. 2005. Roundup, Montana climate summary. Available at: <http://www.wrcc.dri.edu/summary/climsmmt.html>. Accessed March 3, 2005.

Potential and Pitfalls of Prescribed Burning Big Sagebrush Habitat to Enhance Nesting and Early Brood-Rearing Habitats for Greater Sage-Grouse

Jeffrey L. Beck¹, J. Garrett Klein¹, Justin Wright¹, and Kenneth P. Wolfley¹

ABSTRACT

We describe short-term (≤ 10 yrs) and long-term (> 10 yrs) responses of prescribed burning to enhance nesting and early brood-rearing habitat for greater sage-grouse (*Centrocercus urophasianus*). Our primary objective was to provide a literature synthesis to identify short- and long-term responses of prescribed burning to important components of sage-grouse nesting and early brood-rearing habitats in mountain (*Artemisia tridentata* vaseyana) and Wyoming (*A. t. wyomingensis*) big sagebrush. In our synthesis, we evaluated ecological status (bare ground and litter), food availability (forbs and insects), and vegetation structure (grass and sagebrush cover). We used six criteria to identify $n = 12$ papers providing meaningful and rigorous results. Of these papers, six reported the effects of burning in mountain big sagebrush; seven in Wyoming big sagebrush; and one provided information for mountain and Wyoming big sagebrush. Our findings point to some potential for short-term enhancement of forbs and grasses in mountain big sagebrush, but no long-term enhancement of herbs in mountain big sagebrush or short- or long-term enhancement of herbs in Wyoming big sagebrush. In particular, prescribed burning leads to a pronounced negative response in sagebrush cover that lasts for at least a few decades. Based on our findings, we cannot recommend burning Wyoming big sagebrush to enhance sage-grouse nesting or early brood-rearing habitat and we suggest prescribed burning has limited short-term value in enhancing forbs and grasses for sage-grouse inhabiting Mountain big sagebrush.

INTRODUCTION

Three major subspecies of big sagebrush (*Artemisia tridentata*; Beetle 1960; Beetle and Young 1965) dominate the sagebrush biome, a region that historically encompassed $> 600,000$ km² of western North America (Küchler 1970; West 1983; Connelly and others 2004). These subspecies of big sagebrush inhabit areas according to climate, soils, and topography (Tisdale and Hironaka 1981; Bunting and others 1987). Mountain big sagebrush (*A. t. vaseyana*) grows on well-developed soils typically at the highest elevations where cooler temperatures and 31–51 cm of annual precipitation prevail; basin big sagebrush (*A. t. tridentata*) occurs at warmer, lower elevations with 25 to 46 cm of annual precipitation on deep, well-drained soils; and, Wyoming big sagebrush (*A. t. wyomingensis*), the

most extensive complex in the Intermountain West (Tisdale 1994), inhabits shallower, sometimes slightly saline soils in warmer regions than mountain big sagebrush with 18 to 31 cm of annual precipitation (Tisdale and Hironaka 1981; Bunting and others 1987). Due to greater moisture, forb and grass production is typically higher in mountain big sagebrush than in basin or Wyoming big sagebrush communities (Bunting and others 1987).

Because basin, mountain, and Wyoming big sagebrush are not root-sprouting shrubs, they can be effectively controlled with prescribed burning programs (Pechanec and others 1965). The size of a burn and characteristics of the site are important factors affecting reestablishment of sagebrush. Sagebrush must reestablish through seeds that have been dispersed into burns from unburned stands by wind, water erosion, or animals (Meyer 1994). Based on cover and density values from various studies, Baker (2006) approximated post-burn recovery periods to be from 35 to 100 years for mountain big sagebrush and 50 to 120 years for Wyoming big sagebrush. Slow reestablishment of sagebrush provides opportunities for herbaceous plants to establish due to limited competition for water, sunlight, and nutrients.

Throughout the sagebrush biome, sagebrush obligate wildlife species such as greater sage-grouse (*Centrocercus urophasianus*) depend on the structural components (for example grass and shrub cover and height) and functional attributes (for example food availability) of sagebrush communities for food and cover. Within sagebrush/bunchgrass landscapes, successful (in other words, ≥ 1 egg hatched) sage-grouse nests are characterized by cover and height of sagebrush and residual grasses that are greater than randomly available (Sveum and others 1998; Holloran and others 2005). Functional early brood-rearing habitats provide protein-rich insects including ants (Hymenoptera), beetles (Coleoptera), and grasshoppers (Orthoptera) critical for early sage-grouse chick survival (Klebenow and Gray 1968; Peterson 1970; Johnson and Boyce 1990). As summers progress, forbs and then sagebrush leaves form larger proportions of chick and juvenile sage-grouse diets (Klebenow and Gray 1968; Peterson 1970) until the food habits of juveniles parallel that of adult sage-grouse at about 3 months (Rasmussen and Griner 1938). Consequently, the structural and functional characteristics of nesting and early brood-rearing habitats in burned areas are of critical importance to sage-grouse population persistence.

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Jeffrey L. Beck, J. Garrett Klein, Justin Wright, and Kenneth P. Wolfley, Department of Renewable Resources, University of Wyoming

Burning sagebrush was originally considered an important tool to increase desirable herbaceous forage species limited by dense stands of older sagebrush (for example Mueggler and Blaisdell 1959). Range managers recognized that improper burning and livestock grazing after burning in sagebrush communities leads to degraded soil and vegetation conditions (Harniss and Murray 1973) and that prescribed fire can also increase habitat diversity (Britton and others 1981). Numerous studies have demonstrated that prescribed burning has negative effects (for example Fischer and others 1996; Wambolt and others 2001; Beck and others 2009) on sage-grouse habitats in mountain and Wyoming big sagebrush. However, some still suggest burning sage-grouse habitats is appropriate under certain scenarios (for example Crawford and others 2004), yet, these scenarios present risks to sage-grouse habitats and populations (Schroeder and others 2006).

Our objective was to provide a literature synthesis to identify short-term (≤ 10 yrs) and long-term (> 10 yrs) responses (Nelle and others 2000; Crawford and others 2004) of prescribed burning to important components of sage-grouse nesting and early brood-rearing habitats in mountain and Wyoming big sagebrush. Early brood-rearing habitats have been defined as those areas near the nest of hatch used by sage-grouse hens with chicks up to 3 weeks following hatch (Connelly and others 2000). These differ from late brood-rearing habitats, which are habitats used by sage-grouse following desiccation of herbaceous vegetation in sagebrush uplands (Connelly and others 2004). We focus our review on nesting and early brood-rearing habitats because of their overall importance to sage-grouse population productivity and the fact that they occur in similar sagebrush habitats. We selected mountain and Wyoming big sagebrush because they collectively form the majority of available habitats used by sage-grouse; basin big sagebrush and other sagebrush species are also used by sage-grouse, but are more constrained in distribution (Tisdale and Hironaka 1981; Connelly and others 2004). To accomplish our objective we selected four habitat features required by nesting and brood-rearing sage-grouse (forb availability, insect availability, grass cover, and sagebrush canopy cover) and two features indicative of ecological status following burning (bare ground and litter). We selected this suite of features because they are commonly measured in studies evaluating vegetation and wildlife responses to burned big sagebrush communities and because the four grouse habitat features reflect food and cover necessary for reproduction and survival of sage-grouse (Anderson and Gutzwiller 2005).

METHODS

We reviewed articles published in scientific journals to identify short-term (≤ 10 yrs) and long-term (> 10 yrs) effects of prescribed burning on sage-grouse habitat features in Wyoming big and mountain big sagebrush communities. We did not include theses and dissertations in our review to avoid including literature that had not undergone rigorous peer review. We searched scientific literature data bases using key words including fire, insects, mountain big sagebrush, prescribed burning, and Wyoming big sagebrush to identify potential papers for our review.

We reviewed the study design of each paper to ensure results would provide us with information necessary to rigorously summarize short- and long-term effects of prescribed burning on six features of sage-grouse habitats in Wyoming and mountain big sagebrush. To obtain information on the widespread effects of burning in Wyoming and mountain big sagebrush, we included articles from areas irrespective of whether sage-grouse were known to occur in those areas. The specific criteria we used to identify studies suitable for our review included: (1) reported results were only from prescribed burning programs, excluding results from wildfire studies; (2) prescribed burns must be in vegetation that was not seeded before or after the burn to better understand the influence of prescribed burning on systems that were composed of native perennial species; (3) avoiding the potential bias from studies that provided results on habitat features selected by grouse within burned habitats; (4) when mountain and Wyoming big sagebrush occurred together in the same burn area, results were reported by species; (5) each study had to compare responses in prescribed burned areas to unburned controls; and (6) differences in estimates had to be corroborated with results from statistical hypothesis tests (for example paired *t*-test or ANOVA).

To facilitate efficient reporting of our results, we categorized habitat features into ecological status (bare ground and litter), food availability (forb [canopy cover, frequency, relative abundance, or production] and insect abundance), and vegetation structure (grass cover [basal or canopy] and sagebrush canopy cover) categories. In those cases where variable effects were reported from different sites within the same study, we report the majority response. We considered comparisons in sage-grouse habitat features between burned and unburned control areas to be statistically different when $P \leq 0.05$. Consequently, neutral responses indicate no detectable change in the values of habitat features, whereas negative indicates a statistically detectable decrease in the value of a habitat feature and positive indicates a statistically detectable increase in the value of a feature.

Table 1—Literature reviewed to describe the effects of prescribed burning on short (<10 years) and long-term (≥10 years) response of greater sage-grouse habitat in mountain and Wyoming big sagebrush.

Reference ^a	Location ^b	Burn Timing	Mountain Big Sagebrush		Wyoming Big Sagebrush	
			Short-term	Long-term	Short-term	Long-term
Beck and others (2009)	ID	Autumn				X
Cook and others (1994)	WY	Spring	X			
Fischer and others (1996)	ID	Autumn			X	
Holmes (2007)	OR	Autumn	X			
Peek and others (1979)	ID	Autumn			X	
Perryman and others (2002)	CO, UT	Autumn			X	X
Pyle and Crawford (1996)	OR	Autumn and Spring	X			
Seefeldt and others (2007)	ID	Autumn	X			
Van Dyke and Darragh (2006)	MT	Autumn and Spring	X	X		
Wambolt and others (2001)	MT	Autumn and Spring	X	X	X	X
Wambolt and Payne (1986)	MT	Autumn			X	X
Wroblewski and Kauffman (2003)	OR	Autumn			X	

^aWe suggest consulting these references for other short- or long-term responses that we do not consider in our paper.

^bLocations are: CO = Colorado, ID = Idaho, MT = Montana, OR = Oregon, UT = Utah

RESULTS

We identified 12 studies that met our selection criteria (table 1). Of these studies, six provided information on the effects of burning in mountain big sagebrush (six short-term and two long-term) and seven in Wyoming big sagebrush (six short-term and four long-term). One study provided information on the short- and long-term effects of prescribed burning in mountain big sagebrush and Wyoming big sagebrush (table 1).

Short-Term Responses

We did not review any studies that reported short-term responses in bare ground in mountain big sagebrush. One study reported a negative short-term response in litter in mountain big sagebrush. We found no short-term response in litter (2 studies) or bare ground (2 studies) in Wyoming big sagebrush (table 2). Reviewed studies indicated a neutral (3 of 6 studies) or positive (3 of 6 studies) response in forb availability in mountain big sagebrush. All reviewed studies (5/5) reported no short-term response in forb availability in Wyoming big sagebrush; however, one study reported a one-year negative response from burning in availability of two of nine sage-grouse food forb species studied in southeastern Oregon (table 2). One paper each provided information on short-term responses in insect availability in mountain and Wyoming big sagebrush. Short-term response of insects was neutral in mountain big sagebrush and neutral (beetles and grasshoppers) or negative (ants) in Wyoming big sagebrush (table 2). Short-term response of grass cover following prescribed burning in mountain big sagebrush was neutral (3 of 6 studies) or

positive (3 of 6 studies). Short-term response of grass cover in Wyoming big sagebrush was neutral (3 of 4 studies) or positive (1 of 4 studies; table 2). All studies reported negative short-term responses in sagebrush canopy cover following prescribed burning in mountain and Wyoming big sagebrush (table 2).

Long-Term Responses

We did not identify any studies that reported long-term responses in bare ground or litter in mountain big sagebrush. Long-term responses in Wyoming big sagebrush for litter and bare ground were neutral (bare ground, two of three studies; litter, one of two studies) or negative (bare ground, one of three studies; litter, one of two studies; table 2). We identified two studies that reported no long-term response of prescribed burning on forb availability in mountain big sagebrush. Three of four long-term studies in Wyoming big sagebrush reported no long-term response, but one of four studies reported a positive response in forb availability in Wyoming big sagebrush (table 2). No papers summarized long-term responses in insect availability in either mountain or Wyoming big sagebrush (table 2). One study each reported a neutral or positive response in grass cover to prescribed burning in mountain big sagebrush, while four studies indicated a neutral long-term response in grass cover following prescribed burning in Wyoming big sagebrush (table 2). Reviewed studies suggest a pronounced negative long-term effect of prescribed burning on sagebrush canopy cover in mountain (2 studies) and Wyoming big sagebrush (3 studies; table 2).

Table 2—Literature describing the short- (<10 years) and long-term (≥10 years) effects (neutral, positive [+], negative [–]) of prescribed burning on sage-grouse habitat features in mountain and Wyoming big sagebrush.

Habitat Features ^a	Mountain Big Sagebrush			Wyoming Big Sagebrush		
	Neutral	+	–	Neutral	+	–
SHORT-TERM						
Ecological Status						
Bare Ground				6,11		
Litter			4	5,6		
Food Availability						
Forbs ^b	2,8,10	4,7,9		3,5,6,10,12		12
Insects	7			3 ^c		3 ^d
Vegetation Structure						
Grass Cover ^e	7,8,10	2,4,9		5,10,11	6	
Sagebrush Canopy Cover			7,8,9,10			5,10,11
LONG-TERM						
Ecological Status						
Bare Ground				6,11		1
Litter				6		1
Food Availability						
Forbs	9,10			1,6,10	11	
Insects						
Vegetation Structure						
Grass Cover	10	9		1,6,10,11		
Sagebrush Canopy Cover			9,10			1,10,11

^a(1)Beck and others (2009), (2) Cook and others (1994), (3) Fischer and others (1996), (4) Holmes (2007), (5) Peek and others (1979), (6) Perryman and others (2002), (7) Pyle and Crawford (1996), (8) Seefeldt and others (2007), (9) Van Dyke and Darragh (2006), (10) Wambolt and others (2001), (11) Wambolt and Payne (1986), (12) Wroblewski and Kauffman (2003).

^bForb availability was reported as canopy cover, frequency, relative abundance, or production.

^cColeoptera and Orthoptera.

^dHymenoptera.

^eBasal or canopy cover.

DISCUSSION

Our study provides a synthesis of published literature identifying short- and long-term responses of habitat features used by sage-grouse for nesting and early brood-rearing to prescribed burning in mountain and Wyoming big sagebrush. Our findings point to some potential for short-term enhancement of forbs and grasses in mountain big sagebrush, but no long-term enhancement of forbs or grasses in mountain big sagebrush or short- or long-term enhancement of grasses or forbs in Wyoming big sagebrush. In particular, prescribed burning leads to a pronounced negative response in sagebrush cover that lasts for at least a few decades. Of the studies we reviewed, the study conducted by Wambolt and others (2001) spanned the longest time period. They reported that cover of mountain and Wyoming big sagebrush requires more than 16 and 32 years, respectively, to return to preburn levels following prescribed burning. In comparison, several studies have reported slow recovery rates for big sagebrush particularly Wyoming big sagebrush (Baker 2006; Lesica and others 2007; Beck and others 2009).

There is little available evidence that provides support for positive responses of prescribed burning to sage-grouse habitat. There may be opportunities for limited prescribed burning (0.1 ha burns) in dense stands of mountain big sagebrush summer habitats to increase forbs (Pyle and Crawford 1996). Similarly, Dahlgren and others (2006) reported increased brood use within 40.5 ha mountain big sagebrush plots in south-central Utah that had been partially mechanically or chemically treated; however, greatest brood use for all treatments occurred within 10 m of the edge of treatments where sagebrush cover was available, suggesting the importance of treating small patches of sagebrush to invigorate forbs for brood use. However, systematic and routine burning of sagebrush rangelands will not lead to stabilizing and increasing sage-grouse populations (Wambolt and others 2002). Burning sagebrush should only be undertaken after carefully considering sage-grouse habitat requirements and concomitant habitat conditions (Dahlgren and others 2006). Due to the long recovery period for mountain and Wyoming big sagebrush after a burn (Baker

2006), sage-grouse populations could be lost in an area following frequent, large-scale burning (Pederson and others 2003).

CONCLUSIONS

Based on our findings, we cannot recommend burning Wyoming big sagebrush to enhance sage-grouse nesting or early brood-rearing habitat and we suggest small prescribed burns in mountain big sagebrush have limited short-term value in enhancing forbs and grasses for nesting and early brood-rearing sage-grouse. Our recommendation agrees with Schroeder and others' (2006) viewpoint and a recommendation by Beck and others (2009) who argue that loss of sagebrush structural characteristics is so great following prescribed fire that burning may maximize detrimental effects to sagebrush dependent species such as sage-grouse. For instance, burned mountain big sagebrush may not recover sufficiently to support nesting or brood-rearing for more than 20 years (Nelle and others 2000). Available research indicates that burning has some potential to enhance mountain big sagebrush nesting and brood-rearing habitats in small patches, but no potential to enhance habitats used by greater sage-grouse in Wyoming big sagebrush at any scale.

ACKNOWLEDGMENTS

We thank J. W. Connelly for helpful comments that improved our paper.

REFERENCES

- Anderson, S.H., Gutzwiller, K.J. 2005. Wildlife habitat evaluation. In: Braun, C.E., ed. in Techniques for wildlife investigations and management. 6th ed. Bethesda, Maryland: The Wildlife Society: 489–502.
- Baker, W.L. 2006. Fire and restoration of sagebrush ecosystems. *Wildlife Society Bulletin* 34:177–185.
- Beck, J.L., Connelly, J.W., Reese, K.P. 2009. Recovery of greater sage-grouse habitat features in Wyoming big sagebrush following prescribed fire. *Restoration Ecology*. 17:393–403.
- Beetle, A.A. 1960. A study of sagebrush—Section *Tridentatae* of *Artemisia*. *Bull* 368. Laramie: University of Wyoming, Agricultural Experiment Station, 83 pp.
- Beetle, A.A., Young, A. 1965. A third subspecies in the *Artemisia tridentata* complex. *Rhodora* 67:405–406.
- Britton, C.M., Clark, R.G., Sneva, F.A. 1981. Will your sagebrush range burn? *Rangelands* 3:207–208.
- Bunting, S.C., Kilgore, B.M., Bushey, C.L. 1987. Guidelines for prescribed burning sagebrush-grass rangelands in the northern Great Basin. *Gen. Tech. Rep. INT* 231. pp. 33.
- Cook, J.G., Hershey, T.J., Irwin, L.L. 1994. Vegetative response to burning on Wyoming mountain-shrub big game ranges. *Journal of Range Management* 47:296–302.
- Connelly, J.W., Knick, S.T., Schroeder M.A., Stiver, S.J. 2004. Conservation assessment of greater sage-grouse and sagebrush habitats. Western Association of Fish and Wildlife Agencies. Unpublished Report. Cheyenne, Wyoming.
- Connelly, J.W., Schroeder, M.A., Sands, A.R., Braun, C.E. 2000. Guidelines to manage sage grouse populations and their habitats. *Wildlife Society Bulletin* 28:967–985.
- Crawford, J.A., et al. 2004. Ecology and management of sage-grouse and sage-grouse habitat. *Journal of Range Management* 57:2–19.
- Dahlgren, D.K., Chi, R., Messmer, T.A. 2006. Greater sage-grouse response to sagebrush management in Utah. *Wildlife Society Bulletin* 34:975–985.
- Fischer, R.A., Reese, K.P., Connelly, J.W. 1996. An investigation on fire effects within xeric sage grouse habitat. *Journal of Range Management* 49:194–198.
- Harniss, R.O., Murray, R.B. 1973. 30 years of vegetal change following burning of sagebrush-grass range. *Journal of Range Management* 26: 322–325.
- Holloran, M.J. et al. 2005. Greater sage-grouse nesting habitat selection and success in Wyoming. *Journal of Wildlife Management* 69:638–649.
- Holmes, A.L. 2007. Short-term effects of a prescribed burn on songbirds and vegetation in mountain big sagebrush. *Western North American Naturalist* 67:292–298.
- Johnson, G.D., Boyce, M.S. 1990. Feeding trials with insects in the diet of sage-grouse chicks. *The Journal of Wildlife Management*, 54:89–91.
- Lesica, P., Cooper S.V., Kudray, G. 2007. Recovery of big sagebrush following fire in southwest Montana. *Rangeland Ecology and Management* 60:261–269.
- Klebenow, D.A., Gray, G.M. 1968. Food habits of juvenile sage grouse. *Journal of Range Management* 21:80–83.
- Küchler, A.W. 1970. Potential natural vegetation. USDI, Geological Survey, the National Atlas of the United States of America. Washington D.C.: U.S. Government Printing Office: 89–92.
- Meyer, S.E. 1994. Germination and establishment ecology of big sagebrush: implications for community restoration. In: Monsen, S.B., Kitchen, S.G., eds. Ecology and management of annual rangelands symposium, proceedings; 1992 May 18–22; Boise, ID. *Gen. Tech. Rep. INT*-313. Ogden, UT: Intermountain Research Station: 244–251.
- Mueggler, W.F., Blaisdell, J.P. 1959. Effects on associated species of burning, rotobating, spraying, and railing sagebrush. *Journal of Range Management* 11:61–66.
- Nelle, P.J., Reese, K.P., Connelly, J.W. 2000. Long-term effects of fire on sage grouse habitat. *Journal of Range Management* 53:586–591.
- Pechanec, J.F., Plummer, A.P., Robertson, J.H., Hull, A.C., Jr. 1965. Sagebrush control on rangelands. USDA, Handbook Number 277, Washington D.C.
- Pederson, E.K., Connelly, J.W., Hendrickson, J.R., Grant, W.E. 2003. Effect of sheep grazing and fire on sage grouse populations in southeastern Idaho. *Ecological Modelling* 165:23–47.
- Peek, J.M., Riggs, R.A., Lauer, J.L. 1979. Evaluation of fall burning on bighorn sheep winter range. *Journal of Range Management* 32:430–432.
- Perryman, B.L., Olson, R.A., Petersburg, S., Naumann, T. 2002. Vegetation response to prescribed fire in Dinosaur National Monument. *Western North American Naturalist* 62:414–422.
- Peterson, J.G. 1970. The food habits and summer distribution of juvenile sage grouse in central Montana. *Journal of Wildlife Management* 34: 147–155.
- Pyle, W.H., Crawford, J.A. 1996. Availability of foods of sage grouse chicks following prescribed fire in sagebrush-bitterbrush. *Journal of Range Management* 49:320–324.
- Rasmussen, D.I., Griner, L.A. 1938. Life history and management studies of the sage grouse in Utah, with special reference to nesting and feeding habits. *Transactions of North American Wildlife Conference* 3:852–864.
- Schroeder, M.A. et al. 2006. Society for Range Management issue paper: ecology and management of sage-grouse and sage-grouse habitat-a reply. *Rangelands* 28(3):3–7.
- Seefeldt, S.S., Germino, M., DiCristina, K. 2007. Prescribed fires in *Artemisia tridentata* ssp. *vaseyana* steppe have minor and transient effects on vegetation cover and composition. *Applied Vegetation Science* 10:249–256.
- Sveum, C.M., Edge, W.D., Crawford, J.A. 1998. Nesting habitat selection by sage grouse in south-central Washington. *Journal of Range Management* 51:265–269.
- Tisdale, E.W. 1994. Wyoming big sagebrush SRM 403. In: Shiflet, T.N., ed. *Rangeland cover types of the United States*. Denver, Colorado: Society for Range Management: 42–43.
- Tisdale, E.W., Hironaka, M. 1981. The sagebrush-grass ecoregion: a review of the ecological literature. *Forest, Wildlife, and Range Experiment Station, Bulletin No. 33 (Contribution No. 209)*. University of Idaho, Moscow.

- Van Dyke, F., Darragh, J.A. 2006. Short- and long-term changes in elk use and forage production in sagebrush communities following prescribed burning. *Biodiversity and Conservation* 15:4375–4398.
- Wambolt, C.L. et al. 2002. Conservation of greater sage-grouse on public lands in the western U.S.: implications of recovery and management policies. Policy Analysis Center for Western Public Lands. Policy Paper SG-02-02, Caldwell, Idaho.
- Wambolt, C.L., Payne, G.F. 1986. An 18-year comparison of control methods for Wyoming big sagebrush in southwestern Montana. *Journal of Range Management* 39:314–319.
- Wambolt, C.L., Walhof, K.S., Frisina, M.R. 2001. Recovery of big sagebrush communities after burning in south-western Montana. *Journal of Environmental Management* 61:243–252.
- West, N.E. 1983. Western Intermountain sagebrush steppe. *In*: West, N.E., ed. *Temperate deserts and semi-deserts. Ecosystems of the World*, Volume 5. Amsterdam, Netherlands: Elsevier Scientific Publishing Company: 351–374.
- Wroblewski, D.W., Kauffman, J.B. 2003. Initial effects of prescribed fire on morphology, abundance, and phenology of forbs in big sagebrush communities in southeastern Oregon. *Restoration Ecology* 11:82–90.

Response of Obligate Birds to Mechanical Manipulations in a Sagebrush Ecosystem Near Gunnison, Colorado

Patrick A. Magee¹, Jason Brooks¹, Nick Hirsch¹, and Tyler L. Hicks¹

ABSTRACT

Sagebrush ecosystems across western North America have declined in productivity and biodiversity with significant declines in sagebrush obligate birds. Mechanical methods to reduce sagebrush canopy cover and encourage herbaceous undergrowth have been implemented to restore sagebrush ecosystems. How these treatments affect sagebrush obligate birds has not been documented. In this study, we hypothesized that nesting success would decline in mechanically treated (mowed and dixie harrowed) plots immediately and for two years after treatment. Twelve 6-ha plots within the South Parlin Common Allotment in the Gunnison Basin, Colorado were established in 2005. Four replicates of untreated (control), mowed and disked (dixie harrow) plots were sampled in 2005 (pre-treatment) and in 2006 and 2007. We located nests using the systematic walk and incidental flush method. We revisited nests every three to five days to determine nest fate. We located 142 nests of nine species in 90 nest searches and expended over 600 man hours. Mayfield nest success (proportion of nests that successfully hatch at least one young) and clutch size were similar among treatments. Nest success of artificial nests also was similar among treatments. Predation was the leading cause of nest failure accounting for 71 to 77 percent of all nest failures and small mammals were implicated in 76 percent of the depredations. Least chipmunks may have been the principle predator of sagebrush bird nests in our study. Small scale mechanical treatments to restore sagebrush apparently do not negatively affect sagebrush birds within two years post-treatment, but longer duration studies and larger sample sizes are required to better assess the impact of treatments on sagebrush avifauna.

INTRODUCTION

Sagebrush ecosystems across the western United States have been declining for several decades in terms of habitat quantity and quality, leading some scientists to suggest the sagebrush steppe ecosystem is one of the most endangered ecosystems in North America (Noss and others 1995; Mac and others 1998). Avian ecologists have described the ecosystem as one “teetering on the edge of collapse” (Knick and others 2003).

The avifauna indicates that the system is incapable of supporting historic population densities. For example, 63 percent of shrubland species in western North America declined from the mid 1960s to the late 1990s (Paige

and Ritter 1999). According to one summary report of shrubsteppe landscapes, 16 of 25 upland avian species within these systems have experienced significant declines (Dobkin and Sauder 2004). The two species of sage-grouse have exhibited dramatic population declines over the last 30 years. The Gunnison sage-grouse (*Centrocercus minimus*) has declined by more than 50 percent, has a small population size, and a restricted range, and therefore is considered highly imperiled (Dunn and others 2005) and globally endangered (IUCN 2007). Less is known about the approximately 100 other bird species that occupy sagebrush habitats (Braun and others 1976). Breeding Bird Surveys (BBS) from 1966 to 1996 indicate the Brewer’s sparrow (*Spizella breweri*) declined by 3.7 percent annually or more than 50 percent in three decades (Colorado Partners in Flight Land Bird Conservation Plan 2000). In addition, lark sparrows (*Chondestes grammacus*) experienced a similar rate of decline, while vesper sparrows (*Pooecetes gramineus*), lark buntings (*Calamospiza melanocorys*), rock wrens (*Salpinctes obsoletus*), horned larks (*Eremophila alpestris*), and common nighthawks (*Chordeiles minor*) -- all common sagebrush species -- declined from 15 to 49 percent from the mid 1960s to present (Dunn and others 2005). The green-tailed towhee (*Pipilo chlorurus*) and the sage sparrow (*Amphispiza belli*), both highly dependent on shrub ecosystems, are species of concern because of their high vulnerability in a landscape that has declined overall. In addition, their population trends are largely unknown due to data gaps (Dobkin and Sauder 2004; Dunn and others 2005). These trends in avian populations coincide with the alteration of over 60 percent of the sagebrush steppe ecosystem through exotic species invasions and conversion to non-native annual grasses. Of the 63 million ha of sagebrush in western North America, little of it has remained unaltered by human activities (Paige and Ritter 1999; Dobkin and Sauder 2004). Over 90 percent of sagebrush streams and springs have been altered causing a hydrological collapse within the sagebrush ecosystem (Dobkin and Sauder 2004). Often, the seeps and springs within the sagebrush are centers for avian biodiversity largely because of the mesic condition of the habitat and the complex vegetation structure (Doyle 2003; Sada and others 2001).

Over much of the sagebrush biome, land managers have implemented habitat treatments in an effort to improve sage-grouse habitat (Dahlgren and others 2006; Sedgwick 2004). Historically, sagebrush “improvement” meant reducing sagebrush cover to promote forage for livestock (Braun and others 1976). In the Gunnison Basin, treatments

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Patrick A. Magee, Jason Brooks, Nick Hirsch, and Tyler L. Hicks, Department of Natural and Environmental Sciences, Western State College of Colorado, Gunnison, CO

are currently applied to reduce sagebrush cover and simultaneously improve the health of the sagebrush ecosystem. Across the sagebrush biome many areas have been invaded by pinon-juniper or non-native annual grasses, or have successional developed into mature sagebrush with little herbaceous understory. The treatments frequently involve mechanical methods that reduce sagebrush cover and encourage growth of herbaceous understory vegetation essential for sage-grouse nesting and early brood-rearing habitat. These projects are designed to benefit sage-grouse, however, little information has been gathered to assess the effects of these treatments on sage-grouse populations, other animals inhabiting sagebrush communities, or the general health of sagebrush ecosystems. Therefore, this study was designed in conjunction with sage-grouse habitat improvement treatments implemented by the Bureau of Land Management (BLM) Gunnison Field Office. The purpose of this project is to describe short-term avifaunal responses (primarily passerines) in sagebrush habitats that have been mechanically treated to improve sage-grouse nesting habitat by reducing sagebrush cover. Specifically we addressed the question of how reproductive success of sagebrush bird species was influenced by mechanically treating sagebrush in the Gunnison Basin, Colorado. We predicted that reproductive success would decline in treated habitats as a result of increased fragmentation that would encourage nest parasitism and predation.

STUDY AREA

The study was conducted in Major Land Resource Area (MLRA) 048-A and Bird Conservation Region (BCR) 16 – the Southern Rocky Mountains. More specifically, research occurred in the South Parlin Flats Common Allotment on BLM lands approximately 20 km southeast of Gunnison, Colorado, south of U.S. Highway 50 and east of Colorado Highway 114. All sampling was restricted to the Sage Hen Pasture which was not grazed by livestock from 2005 through 2007. Sage Hen Pasture is 1,478 ha (3,650 ac) and ranges in elevation from 2,500 to 2,600 m m.s.l. (8,200 to 8,500 feet m.s.l.). The pasture is dominated by Wyoming Big Sagebrush (*Artemisia tridentata wyomingensis*) and other varieties of *Artemisia*. The BLM had previously mapped the pasture to delineate ecological types according to The Ecological Types of the Upper Gunnison Basin (Johnston and others 2001). The three primary ecological types in the Sage Hen Pasture include 1) Wyoming big sagebrush/Indian ricegrass (*Oryzopsis hymenoides*) – SB1, 2) Big sagebrush/muttongrass (*Poa fendleriana*) – SS1, and 3) Antelope bitterbrush (*Purshia tridentata*)-sagebrush/needlegrass (*Acnatherum* spp.) – SS2. Our study primarily focused on the first two ecological types. Soils were deep and well-drained and primarily classified as Parlin-Hopkins channery loams (PhF) and Parlin-Mergel gravelly loams (PmF). The climate in the Gunnison Basin consists of cold winters with moderate snowfall and dry

springs with a late summer monsoon delivering about 40 percent of the annual precipitation. Snow accumulation in winter is important in dictating soil moisture conditions in spring and early summer. The long-term annual precipitation is 26.3 cm (10.36 inches) compared to the 2005 to 2007 mean of 29.0 cm (11.40 inches). Precipitation varied from 24.1 cm (9.49 inches) in 2007 to 32.8 cm (12.91 inches) in 2005. Long-term mean temperature in Gunnison is 37.5 F and the 2005 to 2007 mean was 37.7 F and ranged from 36.7 F in 2006 to 38.8 F in 2007. Generally, 2005 was a relatively wet year with average temperatures, 2006 was slightly wet with below average temperatures, and 2007 was dry with above average temperatures. During May and June only, 2005 was slightly dry with average temperatures, 2006 was relatively wet with above average temperatures, and 2007 was dry with above average temperatures. The three year period averaged 10 cm (4 inches) per year below mean snowfall with the 2006 to 2007 winter approaching half (26.5 cm) the long-term mean snowfall of 49.9 cm (Climate data obtained from: Western Region Climate Center, <http://www.wrcc.dri.edu/summary/climsmco.html>).

METHODS

Within the Sage Hen Pasture, twelve 6-ha (300 m x 200 m) sampling plots were established for avian monitoring. Plot sizes varied slightly as a result of landscape and ecological type variations and as a result of small errors in establishing plot borders in the field. Control and treatment plots were established and stratified based on ecological type (Johnston and others 2001) (figure 1). An effort was made to locate all plots within the same ecological type, but some variation occurred within and among plots; plots were located randomly and were selected based on available habitat within the Sage Hen Pasture study area. Four control plots were untreated, and the other eight plots were treated in October or November 2005. Four plots were treated with a brush mower that mowed the sagebrush to a height of 20 to 30 cm (8 to 12 inches) and minimally disturbed the soil. Four plots were treated with a dixie harrow that raked the soil and thinned the sagebrush by uprooting individual plants. Approximately 33 percent of each of the treated plots was manipulated; treatments were applied in a serpentine pattern to avoid uniform patches of disturbed sagebrush. In addition to mechanical manipulation, the plots were seeded with a native uplands grass/forb mixture including Sandberg bluegrass (*Poa sandbergii*), bottlebrush squirreltail (*Elymus elmoides*), western wheatgrass (*Pascopyrum smithii*), Indian ricegrass (*Oryzopsis hymenoides*), alfalfa (*Medicago sativa*), small burnet (*Sanguisorba minor*), sulfur buckwheat (*Eriogonum umbellatum*), and Lewis flax (*Linum perenne* ssp. *lewisii*).

Sampling occurred in 2005 (pre-treatment) and in 2006 and 2007 (post-treatment).

Nest Searches

Nest searching occurred on all plots following the U.S.D.A. Forest Service protocol (Ralph and others 1993) and the systematic walking and incidental flush methods described by Winter and others (2003). In 2005, all plots were searched once and half of the plots were searched twice. Three rounds of nest searching occurred in 2006 and 2007 in all plots. Plots were searched in late May (after 20 May), early June (before 10 June) and in late June (after June 20). Because of the reduced effort and minimal success in locating nests, the 2005 data did not offer a legitimate temporal control. Nests were primarily located by flushing birds from nest sites. In addition, observations of females carrying nesting materials, females moving toward their nest, and males and females feeding chicks (Ralph and others 1993) were used to locate nests. All nest locations were geo-referenced using a Garmin eTrex Vista GPS unit (NAD83 unit system). The species, number of eggs or chicks, nest height, shrub height, species of shrub, percent of live shrub, distance to treatment or roads, and a general description of each nest site were recorded. Photos of each nest shrub and nest were taken. All located nests were rechecked at approximately 3 day intervals. During each site visit, nest status (active, abandoned, depredated, post-fledged) was recorded along with the number of eggs and/or chicks. Nests were monitored until each nest attempt was completed (ending in success or failure).

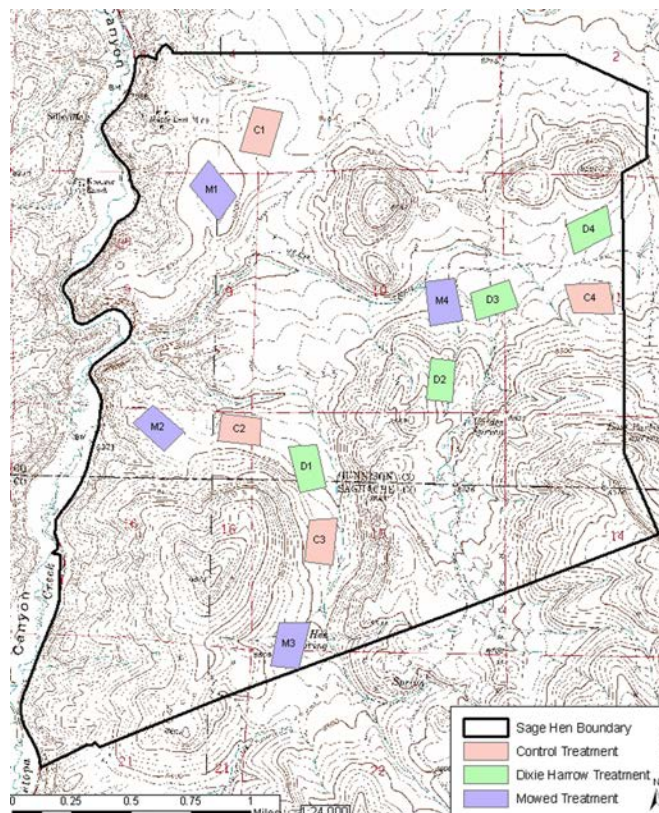


Figure 1—Location and treatment allocation of plots within Sage Hen Pasture in the South Parlin Flats Common Allotment southeast of Gunnison, CO.

Nest Success

We estimated nest success as the probability that a nesting attempt resulted in production of one or more chicks. We focused on this measurement of nest success because our data were less definitive for fledging success (the proportion of nests that produce at least one young that fledges). It is difficult to determine actual nest fate during the transition of nest from nestling to fledged because when we returned to nests after three days and found an empty nest, the nest may have fledged or may have been depredated. We minimized trips to the nest (every three days) to reduce potential researcher impacts. Traditionally, most researchers calculated apparent nest success by dividing the number of successful nests by the total number of nests. However, apparent nest success is biased when applied to samples of clutches that are of different ages (Mayfield 1961; Shaffer 2004). Older clutches are more likely to hatch than the younger clutches as a result of the shorter interval between discovery and hatching (Mayfield 1961, 1975). The longer a nest is exposed to predators and other destructive elements, the more vulnerable it is to nest failure. To avoid bias in success, we used the Mayfield nest success estimator (Mayfield 1961, 1975; Shaffer 2004). This method estimates daily survival by incorporating age of the nest and, therefore, how long nests were exposed to risk while being monitored.

The Mayfield estimator (Mayfield 1961) is calculated using the following equation:

$$P = (1 - N_u/E)^h$$

where,

P = probability of nest success

N_u = number of unsuccessful nests

E = sum of exposure days (number of days from hatch to fate)

h = average incubation period

Mayfield estimates were generated for all species combined and for each species individually. Further, we estimated nest success by treatment type for all species combined and for species with larger sample sizes (Brewer's sparrows, vesper sparrows, and green-tailed towhees). We tested the null hypotheses that there was no difference in clutch size or nest success among the control, mowed, and dixie harrowed plots. We used a non-parametric Kruskal-Wallis test after determining that the data were non-normally distributed and had unequal variances. Our rejection level (α) was 0.05. We also calculated post-hoc power analysis ($1 - \beta$) for a two-sample test (Kapadia and others 2005; Jim zumBrunnen, Colorado State University Graybill Statistical Laboratory, personal communication) using the following equation:

$$Z_\beta = Z_{1-\alpha/2} - \text{ABS}[(\mu_1 - \mu_2) / (\text{SQRT}(2 * \text{Variance}/n))]$$

The β value (or probability of making a type II error) was determined from the normal distribution and the power of the test ($1 - \beta$) was determined by subtraction.

Other indicators of reproductive success included clutch size, egg success, hatch success, and fledge success. The fledge fate was approximated based on the length of the nestling period, but for most nests the final observation was an empty nest with no sure way to determine whether the nest successfully fledged young or was depredated during the last three day interval. If nests were empty before they were expected to fledge, they were assumed depredated. Fledge success was not corrected for exposure and was used only as a comparison with nest success.

Artificial Nests

To enhance our understanding of avian nest success and the mechanism of nest loss, we employed artificial nests. We used 5 cm deep by 10 cm wide wicker woven canary nests and placed one handmade 2.5 cm long blue clay egg in each nest attached by a 3 to 5 cm strand of fishing line to prevent removal of the egg from the nest. Additionally, we added a Chinese quail egg to each nest. The nests were sprayed with a human scent remover prior to placement in the field.

In 2006, in each of the twelve plots, we placed 20 artificial nests along two longitudinal transects. Nests were placed alternatively on the ground and at a height of 20 cm in a shrub. Nest locations and heights were chosen to mimic natural placement by Brewer's sparrows and vesper sparrows. Artificial nests were initially placed on 26 June 2006 in two plots and additional nests were placed on subsequent days in the remaining plots. Each artificial nest was checked at 7 days and 14 days for signs of disturbance. Nest disturbances included both physical (wind, rain) and animal. Animals that visited nests were documented by examining teeth and bill marks in clay eggs, remains of quail eggs, and by looking for additional signs around the nest (tracks, fur/feathers, scat). Nest fate was classified as undisturbed, unknown disturbance, avian predator, small mammal predator, and mid-sized mammal predator. The unknown disturbance category included nests that were disturbed but the cause of the disturbance was not determined; this category potentially included depredations where no predator signs were detected or physical disturbances to the nests occurred.

In addition to observations of artificial eggs at nest sites, we also used eight Cuddeback digital cameras (Non Typical, Inc., Park Falls, WI, cuddebackdigital.com) to detect predators at artificial nests. Three of the cameras were placed in a control plot, three in a dixie harrowed plot and

two cameras in a mowed plot. In complete darkness the cameras were capable of capturing images by using infrared emitters to illuminate the objects when lasers were triggered. However, no night depredation events were recorded.

In 2007, we placed 120 artificial nests in all twelve plots, with 10 nests randomly placed in shrub locations within the plots (no ground nests were used in 2007). The randomly positioned nests helped to reduce the potential for nest predators to systematically locate a series of artificial nests located along a transect. The nests were placed in the field in mid-June and were monitored as in 2006. Twenty Brewer's sparrow nests that were located in 2006, were collected at the end of the 2006 season and were deployed as artificial nests in 2007. These nests were used to compare nest success of wicker nests with real Brewer's sparrow nests. Five old Brewer's sparrow nests were placed in each of the four control plots. Five Casio Model SY-30B infrared remote video cameras were used to monitor artificial nests in 2007. These were used instead of the Cuddeback digital cameras, which did not successfully capture any predators at nests in 2006.

We tested the null hypothesis that there was no difference in nest success of the artificial nests among the treatments using a non-parametric Kruskal-Wallis test with a rejection level of 0.05.

RESULTS

Sagebrush Bird Nest Locations

A total of 142 nests were located within the twelve sample plots during the three year study. We conducted 18 nest searches in 2005 and 36 each in 2006 and 2007. Of the total nests, 21 were found in 2005, 42 in 2006, and 79 in 2007. Ten nests had unknown fates and were not used in the analyses. Nine species accounted for all of the nests with Brewer's sparrows dominating (61 nests), followed by vesper sparrows (44 nests) and green-tailed towhees (16 nests). Other nesting species within the plots included sage thrasher, mourning dove, sage sparrow, common nighthawk, Gunnison sage-grouse, and mallard (table 1). Two horned lark nests were found adjacent to plot boundaries and nests of Clark's nutcracker, northern flicker, common raven, and American kestrel were located within the study pasture in small forest patches. No rock wren nests were located despite the prevalence of this bird within the study site. The three sage sparrow nests represent the first documented sage sparrow nests in Gunnison County and perhaps only the second documented case of sage sparrows nesting in Colorado (Andrews and Righter 1992).

Table 1—Nest searching effort and number and species of bird nests found from 2005-2007 in the Sage Hen Pasture in the Gunnison Basin, Colorado. BRSP = Brewer's Sparrow, VESP = Vesper Sparrow, GTTO = Green-tailed Towhee, SATH = Sage Thrasher, MODO = Mourning Dove, SAGS = Sage Sparrow, CONI = Common Nighthawk, GUSG = Gunnison Sage-grouse, and MALL = Mallard.

Attribute	2005	2006	2007	Total
Nest Searches	18	36	36	90
Initial Nest Search	1 June 2005	21 May 2006	23 May 2007	
Earliest Nest Found	1 June 2005	18 May 2006	21 May 2007	
Calculated Initiation of Nesting	21 May – VESP	29 April – HOLA 5 May – MODO 15 May – BRSP	5 May – SATH 7 May – SAGS 11 May – VESP	
Search Effort	104 hours	256 hours	257.5 hours	617.5 hours
Effort Per Nest	4.9 hours	6.1 hours	3.2 hours	
Total Nests	21	42	79	142
BRSP	6	18	37	61 (43%)
VESP	8	8	28	44 (31%)
GTTO	4	5	7	16 (11%)
SATH	0	3	4	7 (5%)
MODO	0	6	0	6 (4%)
SAGS	1	0	2	3 (2%)
CONI	1	2	0	3 (2%)
GUSG	0	0	1	1 (<1%)
MALL	1	0	0	1 (<1%)

Clutch Size

Green-tailed towhees had the largest overall mean clutch size (3.9 eggs), followed by vesper sparrows and sage thrashers (3.7 eggs). Brewer's sparrows had a mean clutch size of 3.5 eggs. The largest clutch was 5 eggs for sage thrashers and vesper sparrows. Sagebrush birds typically laid 3 or 4 eggs with 4 egg clutches being the most frequent (table 2).

Clutch size did not differ significantly among treatments for the three most abundant nesting birds in the study (Brewer's sparrows, vesper sparrows, and green-tailed towhees). Brewer's sparrows tended to nest in control plots more than mechanically treated plots, whereas the number of vesper sparrow nests was highest in mowed plots. Sixty percent of green-tailed towhee nests were located in dixie harrow plots. No difference in clutch size among treatments was observed for green-tailed towhees and clutch sizes for Brewer's sparrows and vesper sparrows were similar among treatments (table 3).

Reproductive Success

Of the 132 nests, 110 nests (83.3 percent) successfully hatched at least one egg (apparent nest success). Of the 22 unsuccessful nests, 10 (45 percent) were vesper sparrow and 8 (36 percent) were Brewer's sparrow nests. Vesper sparrows had the highest rates of nest loss.

Taking into account exposure time, the overall Mayfield nest success estimate was 71.6 percent (all species and all

years). Brewer's sparrows had the highest Mayfield nest success rate (81.8 percent, $n = 56$), followed by green-tailed towhees at 77.9 percent ($n = 15$). Vesper sparrows had a relatively low nest success rate of 54.2 percent ($n = 40$). The other species had sample sizes lower than 7 and estimates of nest success were unreliable (table 4).

Of the 460 eggs laid by all species in 132 nests, 348 eggs hatched (egg success = 75.6 percent). The percentage of eggs that hatched from nests that were successful serves as an index of egg viability. Overall, egg viability exceeded 90 percent (table 4).

The total nest failure (percentage of nest attempts in which no young were produced) was 38 percent for all species combined. Every common nighthawk and sage thrasher nest produced young, while total nest failure was 16.7 percent for mourning doves, 32.1 percent for Brewer's sparrows, 33.3 percent for sage sparrows, 38.5 percent for green-tailed towhees, and 55.5 percent for vesper sparrows.

Nest Success by Treatment

Overall nest success for all species combined in 2006 and 2007, the two post-treatment years, did not differ by treatment (Kruskal-Wallis $p = 0.672$). Brewer's sparrows had the highest nest success in dixie harrow plots (0.916), whereas vesper sparrows had the highest nest success in control plots (0.703). Nonetheless, differences within species among treatments were not significant (table 5).

Table 2—Comparison of mean clutch size among avian species and across years in the Sage Hen Pasture in the Gunnison Basin, Colorado ($\bar{x}$ = mean clutch size, n = number of nests, range = variability in clutch size among nests). BRSP = Brewer's Sparrow, CONI = Common Nighthawk, GTTO = Green-tailed Towhee, MODO = Mourning Dove, SATH = Sage Thrasher, SAGS = Sage Sparrow, VESP = Vesper Sparrow.

Species	2005 ^a	2006	2007	Overall
BRSP	$\bar{x}$ = 3.5 n = 2 Range = 1 (3-4) ^c	$\bar{x}$ = 3.6 n = 18 Range = 2 (2-4)	$\bar{x}$ = 3.4 n = 32 Range = 2 (2-4)	$\bar{x}$ = 3.5 n = 52 Range = 2 (2-4)
VESP	$\bar{x}$ = 3.75 n = 4 Range = 1 (3-4)	$\bar{x}$ = 3.8 n = 8 Range = 1 (3-4)	$\bar{x}$ = 3.7 n = 24 Range = 2 (3-5)	$\bar{x}$ = 3.7 n = 35 Range = 2 (3-5)
GTTO	$\bar{x}$ = 3.5 n = 2 Range = 1 (3-4)	$\bar{x}$ = 4.0 n = 4 Range = 0 (4)	$\bar{x}$ = 4.0 n = 6 Range = 0 (4)	$\bar{x}$ = 3.9 n = 12 Range = 1 (3-4)
SATH	No Nests	$\bar{x}$ = 3.7 n = 3 Range = 1 (3-4)	$\bar{x}$ = 3.8 n = 4 Range = 2 (3-5)	$\bar{x}$ = 3.7 n = 7 Range = 2 (3-5)
MODO	No Nests	$\bar{x}$ = 2 n = 6 Range = 0 (2)	No Nests	$\bar{x}$ = 2 n = 6 Range = 0 (2)
SAGS	$\bar{x}$ = 3 n = 1 ^b Range = 0 (3)	No Nests	$\bar{x}$ = 3.5 n = 2 Range = 1 (3-4)	$\bar{x}$ = 3.3 n = 3 Range = 1 (3-4)
CONI	$\bar{x}$ = 2 n = 1 ^b Range = 0 (2)	$\bar{x}$ = 2 n = 2 Range = 0 (2)	No Nests	$\bar{x}$ = 2 n = 3 Range = 0 (2)

^aNest monitoring in 2005 was limited and total number of nests with known clutch sizes was small

^bWhere sample size is 1, the clutch sizes are not means

^cRange = maximum clutch size – minimum clutch size (variation in values of clutch size shown in parentheses)

Nest Failure

Of the unsuccessful nests, lack of hatching of any eggs within a nest was due primarily to depredation. In 2006, 71 percent of the unsuccessful nests were depredated, whereas in 2007, 77 percent of unsuccessful nests were depredated. The remaining nests were abandoned or contained unhatched eggs.

In general, sagebrush birds were slightly more vulnerable to nest losses during the nestling period than during the egg stage. For Brewer's sparrows, 13.2 percent of total nests prior to hatching failed, whereas 18.9 percent of active nests after hatching failed. Vesper sparrows lost 27.8 percent of nests in the egg stage and again during the nestling stage. Green-tailed towhees experienced a 15.4 percent loss of clutches during incubation, compared to 23.1 percent of nests lost during nestling. Mourning doves lost one of seven (14.3 percent) nests during the egg stage, but did not lose any nests during the nestling period. Sage sparrows did not lose any nests during incubation and lost one of three nests during the nestling stage.

Fledging Success

Of the 121 total nests with known fledge fate (includes nests that did not hatch at least one egg), 75 nests (62.0 percent) fledged young. Of the 24 total nest failures between hatch and fledging, 96 percent (23 nests) failed due to depredation. The remaining nest was abandoned. In 2006, 32 of the 42 total nests fledged young successfully (76 percent), whereas in 2007, 43 of the 78 total nests fledged young (55 percent).

Artificial Nest Studies

Because there was no difference in the number of exposure days among artificial nests, the apparent nest success estimator was used to determine nest success. Nest success was similar among treatments for artificial nests or between ground and shrub nests (table 6). Nest success of old Brewer's sparrow's nests was higher than commercially produced wicker nests (but not significantly).

Of the 380 artificial nests, 51.3 percent were depredated. Of these 191 depredated nests, 76.4 percent were depredated by small mammals based on examination of clay and Chinese quail eggs. The small mammal whose teeth most often matched the marks on the clay eggs and that was captured in the act of depredation using infrared video cameras was the least chipmunk (*Tamias minimus*).

DISCUSSION

Nest Success of Sagebrush Birds

Nest success of sagebrush birds in the Gunnison Basin, Colorado varied among species but were similar among treatments. Our prediction that sagebrush birds would experience reduced nesting success in the short-term response to mechanical treatments in sagebrush was not supported. However, our post-hoc power analysis revealed low power to protect against Type II Errors (Power = $1 - \beta$ = 0.06). Therefore, if the null hypothesis is rejected (no difference in nesting performance among treatments) we would do so with a 94 percent probability that the null hypothesis was not true. Therefore, the p-values from the Kruskal-Wallis tests should be applied with great caution and conclusions inferred from these data should also be cautionary.

Mechanical treatments were done to limit disturbance within the treated area to approximately one third of the landscape (at relatively small scales – here 6 ha). The remaining sagebrush allowed adequate nesting structure for birds to successfully inhabit these treatment sites. That the fragmentation of the sagebrush by mechanical treatments did not lead to higher nest losses over controls may be explained by the lack of nest parasites in the study area (no brown-headed cowbirds) and the lack of avian predators at the site. Loggerhead shrikes (*Lanius ludovicianus*) have been implicated as a prime avian nest predator of Brewer's sparrows, sage sparrows, and sage thrashers (Reynolds 1979, 1981), however, no loggerhead shrikes were observed in Sagehen Pasture during the three year study period. Further, common ravens (*Corvus corax*) and black-billed magpies (*Pica pica*) were present, but according to our artificial nest data, these birds rarely depredated nests.

Lack of nest parasitism and avian predators may also explain the relatively high nest success observed in this study. For all species combined, the Mayfield nest success estimate was 71.6 percent (table 4). Approximately 82 percent of Brewer's sparrow nests hatched at least one young successfully in our study compared to 18 percent in Nevada and 90 percent in Oregon (Rotenberry and Weins 1989). For vesper sparrows, nest success ranged from 11 percent in Michigan (Krueger 1981) to 37 percent in Iowa (Patterson and Best 1996) to 54 percent in Washington (Jones and Cornely 2002), compared to 54 percent in our study. Sage thrashers in Idaho had a 73 percent nest success (Reynolds and Rich 1978), compared to 100 percent ($n = 7$) in our study. Green-tailed towhees in Arizona had a nest success of 55 percent (Dobbs and others 1998), compared to 78 percent in our study. Higher nest success in the Gunnison Basin may be attributed to overall better ecosystem health of the shrubsteppe here than in other regions of the sagebrush biome.

Birds in the sagebrush ecosystem face high rates of predation on nests primarily by other birds such as loggerhead shrikes, common ravens, and black-billed magpies (Dobbs and others 1998; Jones and Cornely 2002; Martin and Carlson 1998; Reynolds and others 1999; Rotenberry and Wiens 1989; Rotenberry and others 1999). A variety of snakes have also been observed depredating bird nests in the sagebrush, chief among them the gopher snake (*Pituophis melanoleucus*). Western garter snakes were observed occasionally in our study plots, but we did not document depredation by them. Mammals such as skunks, weasels, and ground squirrels are important nest predators (Dobbs and others 1998; Jones and Cornely 2002; Martin and Carlson 1998; Reynolds and others 1999; Rotenberry and others 1999). Townsend's ground squirrel (*Spermophilus townsendii*) has often been cited as the most significant small mammal nest predator (Rotenberry and Weins 1989). The least chipmunk was cited as a potential predator of Brewer's sparrows, sage sparrows, and sage thrashers (Peterson and Best 1987; Rotenberry and Weins 1989) and of rock wrens (Lowther and others 2000). In our study, we captured least chipmunks entering and robbing artificial nests on infrared video. Most of the teeth marks in the clay eggs were attributed to the least chipmunk. Deer mice are probably the most abundant small mammal in the Gunnison Basin – approximately 11 to 40 million estimated in 2004 (Thrift and Magee 2005). Although deer mice have been identified as nest predators (Rogers and others 1997) and recorded on video raiding bird nests (Pietz and Granfors 2000), they were not recorded on video entering and robbing artificial nests in our study. Further, Wyoming (*Spermophilus elegans*) and golden-mantled ground squirrels (*Spermophilus lateralis*) were present, but no direct evidence that either of these species robbed nests or killed birds was observed. During our study, least chipmunks appeared to be the most significant predator of sagebrush bird nests.

Table 3—Comparison of mean clutch size (+ standard error) and sample sizes among treatments for the three most prolific nesting birds in the Sage Hen Pasture study area in the Gunnison Basin, Colorado ($\bar{x}$ = mean clutch size + standard error, n = number of nests, range = variability in clutch size among nests). A non-parametric Kruskal-Wallis statistic was used to test the null hypothesis that clutch size did not differ among treatments. BRSP = Brewer's Sparrow, VESP = Vesper Sparrow, GTTO = Green-tailed Towhee.

Species	Control	Dixie Harrow	Mowed	Kruskal-Wallis
BRSP	$\bar{x} = 3.57 \pm 0.13$ $n = 21$ Range = 2 (2-4) ^a	$\bar{x} = 3.50 \pm 0.15$ $n = 16$ Range = 2 (2-4)	$\bar{x} = 3.40 \pm 0.16$ $n = 15$ Range = 2 (2-4)	$p = 0.314$
GTTO	$\bar{x} = 4.0$ $n = 2$ Range = 0 (4)	$\bar{x} = 4.0$ $n = 6$ Range = 0 (4)	$\bar{x} = 4.0$ $n = 2$ Range = 0 (4)	Sample Size too Small
VESP	$\bar{x} = 3.78 \pm 0.22$ $n = 9$ Range = 2 (3-5)	$\bar{x} = 3.75 \pm 0.16$ $n = 8$ Range = 1 (3-4)	$\bar{x} = 3.64 \pm 0.17$ $n = 14$ Range = 2 (3-5)	$p = 0.952$

^aRange = maximum clutch size – minimum clutch size (variation in values of clutch size shown in parentheses)

Table 4—Mayfield nest success, egg success, egg viability and total nest failure rates overall and for individual sagebrush obligate and near obligate species nesting in the Sage Hen Pasture of the Gunnison Basin, Colorado, 2005 to 2007. Sample size for egg viability includes only the number of successful nests, whereas for total nest failure sample size is the number of nests with known fate, and for other variables sample size is the total number of nests. BRSP = Brewer's Sparrow, CONI = Common Nighthawk, GTTO = Green-tailed Towhee, MODO = Mourning Dove, SATH = Sage Thrasher, SAGS = Sage Sparrow, VESP = Vesper Sparrow.

Species	Sample Size (number of nests)	Mayfield Nest Success Estimate	Egg Success	Egg Viability	Total Nest Failure
Overall – all species combined	132	0.716	75.6% of 460 eggs, 348 hatched	92.7% $n = 110$	38%
SATH	7	1.000	100%	100% $n = 7$	0% $n = 7$
CONI	3	1.000	100%	100% $n = 3$	0% $n = 3$
SAGS	3	1.000	80.0%	80.0% $n = 3$	33.3% $n = 3$
BRSP	56	0.818	82.3%	93.5% $n = 49$	32.1% $n = 53$
GTTO	15	0.779	83.0%	90.0% $n = 13$	38.5% $n = 13$
MODO	6	0.588	83.3%	100% $n = 5$	16.7% $n = 6$
VESP	40	0.542	65.0%	85.6% $n = 30$	55.5% $n = 36$

Table 5—Mayfield nest success estimates for control, dixie harrow, and mowed plots in the Sage Hen Pasture in the Gunnison Basin, Colorado, 2006 and 2007. BRSP = Brewer's Sparrow, VESP = Vesper Sparrow, and GTTO = Green-tailed Towhee.

Species	Control	Dixie Harrow	Mowed	Kruskal-Wallis
All Species (plots pooled) $N = 119$	0.80 ^a	0.79	0.70	
All species (plots unpooled) $N = 12$	0.73	0.81	0.65	$p = 0.672$
Range of Values	0.39 – 1.0	0.54 – 1.0	0.61 – 0.71	
Standard Error	0.136	0.114	0.027	
CV ^b	37.3%	28.3%	8.5%	
Plot 1	C1 = 0.89 ($n = 14$)	D1 = 1.0 ($n = 8$)	M1 = 0.61 ($n = 15$)	
Plot 2	C2 = 1.0 ($n = 4$)	D2 = 0.54 ($n = 10$)	M2 = 0.61 ($n = 7$)	
Plot 3	C3 = 0.39 ($n = 7$)	D3 = 0.70 ($n = 16$)	M3 = 0.71 ($n = 11$)	
Plot 4	C4 = 0.64 ($n = 14$)	D4 = 1.0 ($n = 2$)	M4 = 0.70 ($n = 12$)	
Sample Size	39	36	45	
Mayfield Nest Success for Three Most Abundant Species				
BRSP	0.791 ($n = 22$)	0.916 ($n = 16$)	0.740 ($n = 16$)	$p = 0.529$
VESP	0.703 ($n = 10$)	0.445 ($n = 9$)	0.487 ($n = 17$)	$p = 0.753$
GTTO	0.504 ($n = 3$)	1.00 ($n = 6$)	1.00 ($n = 3$)	n.a.

^aBold numbers representing the Mayfield nest success estimates for all species combined were calculated by pooling all four replicates within each treatment ($N = 119$), the non-bold numbers represent the means of the four treatments ($N = 12$).

^bCV is the co-efficient of variation calculated as the standard deviation divided by the mean multiplied by 100.

Management and Research Implications

Relatively little information is available regarding responses of sagebrush birds to restoration and management efforts in the sagebrush steppe ecosystem. Most studies measuring bird response to alterations of sagebrush were associated with sagebrush treatments intending to “improve” sagebrush for livestock foraging (Braun and others 1976). In general, the result of these large scale burns, spraying operations, or mechanical projects was a decline in shrub

dependent bird species. For example, Brewer's sparrows declined 54 percent the year following heavy application 2,4-D spraying on a 16 ha (40 ac) plot in Montana (Best 1972) and five years post-treatment, after shrubs died, Brewer's sparrows were almost completely absent from the study site (Pyrah and Jorgenson 1974). Similarly, Brewer's sparrows on a 32 ha (80 ac) plot declined 67 percent one year following spraying and 99 percent two years post spraying in Wyoming (Schroeder and Sturges 1975). Sage

sparrows, Brewer's sparrows, and sage thrashers are "decreasers" following large scale sagebrush treatments, with thrashers and probably sage sparrows decreasing more significantly due to their specialization on large sagebrush (thrashers) and large stands of sagebrush (sage sparrows) (Castrale 1982).

In contrast, smaller sagebrush removal projects and prescribed restoration alterations do not necessarily have short-term negative impacts. For example, after a moderate-sized, incomplete burn in Idaho, researchers did not document a decline by sage sparrows or sage thrashers, and while Brewer's sparrows declined two years after the burn, their populations fluctuated and increased four years post-burn (Peterson and Best 1987). Further, when 16 ha (40 ac) sagebrush plots were sprayed with 2,4-D in 30 m wide strips or where herbicide application resulted in sagebrush kill less than 50 percent, no change in number of Brewer's sparrow and vesper sparrow pairs was documented (Best 1972). Heavily manipulated sagebrush sites (mechanical removal of 100, 75, 50, 25 and 0 percent of sagebrush canopy cover in mosaic pattern) at small scale (4.5 ha) did not correlate with a major change in sage sparrow or Brewer's sparrow territory location or size or population density (Wiens and others 1986).

The types of treatments mentioned above mimic the scale and patterns of mechanical treatments that the BLM Gunnison Field Office prescribed for our study. The principle guidelines included small areas, approximately 30 percent of landscape treated, with treatments done in sinuous pattern. The key to successful treatment design is maintaining adequate shrub cover for shrub nesting sagebrush obligates. Further, our study plots were rested from livestock grazing for two years post-treatment. Rest from grazing may have been an important factor in increasing grass cover and concealing nests of ground nesting birds (vesper sparrows). As grazing is resumed, further study could elucidate its effects on nest success of sagebrush obligate birds.

Small scale treatments did not negatively affect sagebrush birds, but are these treatments beneficial to a group of birds that are among the fastest declining birds in North America (Paige and Ritter 1999)? Whereas sagebrush canopy cover declined in our study plots after mechanical treatments, grass cover and forb cover increased (Adam Payton, BLM, personal communication) perhaps creating positive habitat values including cover for ground nesting species (vesper sparrows) and food resources for all the birds. Further, despite reduction in sagebrush cover, at least one vesper sparrow and one sage thrasher nested within the uprooted sagebrush in dixie harrow plots. Increase in forbs within dixie harrow plots was attributed to pre-existing seed sources and not plants seeded after the treatment (Adam Payton, BLM, personal communication). This forb response may have been associated with the soil disturbance

allowing for more efficient seed germination. An increase in forb cover and diversity likely would enhance insect abundance and habitat quality for nesting adults and broods. More information is needed to assess the positive impacts of treatments on sagebrush birds.

Long term research is required to determine the effective response of birds to treatments in the sagebrush. Evidence from other studies suggests that breeding birds may return to sites they had occupied the previous season (site fidelity) prior to implementation of treatments even though habitat may have declined dramatically as a result of the treatment (Wiens and others 1986). Therefore, a time lag in bird response to habitat changes may occur and this time lag can lengthen if offspring imprint on specific nesting sites. In some cases, anthropogenic alterations of habitat or distribution of predators, nest parasites, or competitors may create habitat traps, where birds continue to select habitat based on structure of vegetation, but preferred nest sites result in lowered nest success. For example, sage sparrows chose nest sites where they had lower nesting success as a result of human mediated redistribution of snakes (Misenhelter and Rotenberry 2000).

Obtaining adequate data to estimate nest success requires that the study plots are large enough to harbor statistically reasonable sample sizes of nesting birds. In our study, plot size was 6 ha (15 ac). The number of nests per plot ranged from 0 to 10 within a season and most plots had 0 to 2 nests of each species. Spatial scale of treatments was marginally adequate for Brewer's sparrows and possibly vesper sparrows, but too small to measure a population level response by most species (Wiens and others 1986). Future studies should be implemented at larger geographical scales, but maintain the ratio of one third or less treated landscape and avoid large blocks of treated sagebrush within areas.

Predation was the major cause of nest failure (71 to 77 percent of nests) in our study and in numerous other sagebrush bird nesting studies (Dobbs and others 1998; Jones and Cornely 2002; Martin and Carlson 1998; Reynolds and others 1999; Rotenberry and others 1999). Predation rates in our study were lower than experienced in several other studies (Dobbs and others 1998; Jones and Cornely 2002; Reynolds 1981; Reynolds and others 1999; Rotenberry and others 1999). Primary predators are not large and medium sized mammals, and probably not avian predators such as corvids and shrikes, but mostly rodents. In this study we were able to directly implicate least chipmunks using video, but much of our evidence relies on indirect sign at the nest which is a problematic method for determining nest predators (Lariviere 1999). More research using remote infrared camera surveillance of real nests is required to determine the relative effect of a variety of predators.

In addition, despite the widespread documentation of predation as the key mortality agent in sagebrush songbirds, it is important to understand predation as a consequence of habitat degradation and alteration. Further, predation is not uniform across time or space within the sagebrush ecosystem and may be influenced greatly by precipitation, episodic predator densities (for example, ground squirrels; Rotenberry and Wiens 1989), and avian diseases, among other complicating factors.

ACKNOWLEDGEMENTS

We thank the BLM Gunnison Field Office, especially Sandy Borthwick, for funding the project through a grant received from the BLM Application of Science Program.

Further, Sandy Borthwick, Russ Japuntich, Sandy Hayes, and Art Hayes contributed to the design, logistics and field work for the project. Further funding was provided by the Western State College Foundation Thornton Fund. The Western State College Department of Natural and Environmental Sciences also supported the research. Special thanks to field assistance provided by Dr. Greg Schrott, Lynn Fleury, Cynthia Baranek, Connor Jandreau, and Ryan Wilbur, as well as Sidra Blake, Dr. Jeff Beck, Russ Japuntich, Adam Payton, Melody de los Santos, and numerous other volunteers with the BLM and Western State College of Colorado. Jessica Young and Sally Thode provided insightful comments and editorial suggestions on the manuscript.

Table 6—Mean + SD nest success for artificial nests in relation to mechanical treatments in the sagebrush ecosystem in the Gunnison Basin, Colorado 2006 to 2007. Across rows represent treatment comparisons, whereas down columns represent artificial nest type comparisons. At bottom of table the Mayfield nest success estimates are repeated for comparison with artificial nests.

Nest Type	Control	Dixie Harrow	Mowed	Kruskal-Wallis
2006 Ground Nests	42.5 ± 20.6%	45.0 ± 30.0%	25.0 ± 28.9%	$p = 0.384$
2006 Shrub Nests	57.5 ± 27.5%	67.5 ± 18.9%	55.0 ± 37.0%	$p = 0.792$
2007 Shrub Nests	37.5 ± 20.6%	42.5 ± 23.6%	62.5 ± 35.9%	$p = 0.282$
2007 Old Brewer's Sparrow Nests	60.0 ± 28.3%	na	na	
Kruskal-Wallis	$p = 0.489$	$p = 0.244$	$p = 0.155$	
Mayfield Nest Success	73 ± 13.6%	81 ± 11.4%	65 ± 2.7%	$p = 0.672$

REFERENCES

- Andrews, R., Righter, R. 1992. Colorado Birds. Denver Museum of Nature and Science. 442 p.
- Best, L.B. 1972. First-year effects of sagebrush control on two sparrows. *Journal of Wildlife Management* 36(2):534-544.
- Braun, C., Baker, E., Maurice, F., Eng, R.L., Gashwiler, J.S., Schroeder, M.H. 1976. Conservation Committee report on effects of alteration of sagebrush communities on the associated avifauna. *Wilson Bulletin* 88(1):165-171.
- Castrale, J.S. 1982. Effects of two sagebrush control methods on nongame birds. *Journal of Wildlife Management* 46(4):945-952.
- Colorado Partners in Flight. 2000. Colorado Land Bird Conservation Plan. [Online]. Available: <http://www.rmbo.org/pif/bcp/phy62/sage/grsg.htm> [May 23, 2008].
- Dahlgren, D.K., Renee, C., Messmer, T.A. 2006. Greater Sage-grouse response to sagebrush management in Utah. *Wildlife Society Bulletin* 34(4):975-985.
- Dobkin, D.S., Sauder, J.D. 2004. Shrubsteppe landscapes in jeopardy. Distributions, abundances, and the uncertain future of birds and small mammals in the Intermountain West. High Desert Ecological Research Institute, Bend, OR. 194 p.
- Dobbs, R.C., Martin, P.R., Martin, T.E. 1998. Green-tailed Towhee (*Pipilo chlorurus*). In *The Birds of North America*, No. 368 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA. 24 p.
- Doyle, G. 2003. Survey of selected seeps and springs within the Bureau of Land Management's Gunnison Field Office Management Area (Gunnison and Saguache Counties, CO). Colorado Natural Heritage Program, Fort Collins, CO. 143 pp.
- Dunn, E.H. et al. 2005. High priority needs for range-wide monitoring of North American landbirds. Partners in Flight Technical Series No. 2.
- Partners in Flight website: <http://partnersinflight.org/pubs/ts/02-MonitoringNeeds.pdf>.
- Hunter, W., Spears, R., Clayton F. 1975. Soil Survey of the Gunnison Area, Colorado: parts of Gunnison, Hindsdale and Saguache Counties. USDA Soil Conservation Service and Colorado Agricultural Experiment Station.
- International Union for the Conservation of Nature (IUCN). 2007. IUCN Red List of Threatened Species. [Online]. <http://www.iucnredlist.org> [June 28, 2008].
- Johnston, B.C., Huckaby, L., Hughes, T.J., Pecor, J. 2001. Ecological types of the Upper Gunnison Basin. Denver (CO): USDA Forest Service, Rocky Mountain Region.
- Jones, S.L., Cornely, J.E. 2002. Vesper Sparrow (*Poocetes gramineus*). In *The Birds of North America*, No. 624 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.
- Kapadia, A.S., Chan, W., Moye, L. 2005. Mathematical statistics with applications. Chapman Hall/CRC, Boca Raton, FL. 617 p.
- Knick, S.T., Dobkin, D.S., Rotenberry, J.T., Schroeder, M.A., Vander H., Mathew, W., van Riper III, C. 2003. Teetering on the edge or too late? Conservation and research issues for avifauna of sagebrush habitats. *Condor* 105: 611-634.
- Krueger, H.O. 1981. Breeding adaptations of the Vesper Sparrow (*Poocetes gramineus*) in a fire-altered ecosystem. M.S. Thesis, Central Michigan University, Mount Pleasant, MI.
- Lariviere, S. 1999. Reasons why predators cannot be inferred from nest remains. *Condor* 101:718-721.
- Lowther, P.E., Kroodsmas, D.E., Farley, G.H. 2000. Rock Wren (*Salpinctes obsoletus*). In *The Birds of North America*, No. 486 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA. 20 p.
- Mac, M.J., Opler, P.A., Haecker, E.P., Doran, P.D. 1998. Status and trends of the nation's biological resources. Vol. 2. USDI, US Geological Survey, Reston, VA.

- Martin, J.W., Carlson, B.A. 1998. Sage Sparrow (*Amphispiza belli*). In The Birds of North America, No. 326 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA. 20 p.
- Mayfield, H.F. 1961. Nesting success calculated from exposure. Wilson Bulletin 73:255-261.
- Mayfield, H.F. 1975. Suggestions for calculating nest success. Wilson Bulletin 86(4):456-466.
- Misenhelter, M.D., Rotenberry, J.T. 2000. Choices and consequences of habitat occupancy and nest site selection in sage sparrows. Ecology 81(10):2892-2901.
- Noss, R.F., LaRoe III, E.T., Scott, J.M. 1995. Endangered Ecosystems of the United States: A preliminary assessment of loss and degradation. USDI National Biological Service. Biological report 28. Washington, D.C. 58 p.
- Paige, C., Ritter, S.A. 1999. Birds in a sagebrush sea: managing sagebrush habitats for bird communities. Partners in Flight Western Working Group, Boise, ID. 47 p.
- Patterson, M.P., Best, L.B. 1996. Bird abundance and nesting success in Iowa CRP fields: the importance of vegetation structure and composition. American Midland Naturalist 135:153-176.
- Peterson, K.L., Best, L.B. 1987. Effects of prescribed burning on nongame birds in a sagebrush community. Wildlife Society Bulletin 15:317-329.
- Pietz, P.J., Danfors, D.A. 2000. Identifying predators and fates of grassland passerine nests using miniature video cameras. Journal of Wildlife Management 64(1):71-87.
- Pyrh, D., Jorgenson, H.E. 1974. Ecology of sagebrush control. Montana Fish and Game Department. Job Report, Federal Aid Project W-105-R-9.
- Ralph, C.J., Geupel, G.R., Pyle, P., Martin, T.E., DeSante, D.F. 1993. Handbook of Field Methods for Monitoring Landbirds. Gen. Tech. Rep. PSW-GTR-144. Albany, CA: Pacific Southwest Research Station, Forest Service, USDA 41 p.
- Reynolds, T.D. 1979. The impact of loggerhead shrikes on nesting birds in a sagebrush environment. Auk 96:798-800.
- Reynolds, T.D. 1981. Nesting of the sage thrasher, and Brewer's sparrow in southeastern Idaho. Condor 83:61-64.
- Reynolds, T.D., Rich, T.D. 1978. Reproductive ecology of the sage thrasher (*Oreoscoptes montanus*) on the Snake River Plain in south-central Idaho. Auk 95:580-582.
- Reynolds, T.D., Rich, T.D., Stephens, D.A. 1999. Sage Thrasher (*Oreoscoptes montanus*). In The Birds of North America, No. 463 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA. 24 p.
- Rogers, C.M., Taitt, M.J., Smith, J.N.M., Jongeian, G. 1997. Nest predation and cowbird parasitism create a demographic sink in wetland-breeding song sparrows. Condor 99:622-633.
- Rotenberry, J.T., Wiens, J.A. 1989. Reproductive biology of shrubsteppe passerine birds: geographical and temporal variation in clutch size, brood size, and fledging success. Condor 91(1): 1-13.
- Rotenberry, J.T., Patten, M.A., Preston, K.L. 1999. Brewer's Sparrow (*Spizella brewerii*). In The Birds of North America, No. 390 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA. 24 p.
- Sada, D.W., Williams, J.E., Silvy, J.C., Halford, A., Ramakka J., Summers, P., Lewis, L. 2001. Riparian area management: A guide to managing restoring, and conserving springs in the western US. Technical Reference 1737-17. Bureau of Land Management, Denver, CO. BLM/ST/ST-01/001+1737. 70 p.
- Schroeder, M.H., Sturges, D.L. 1975. The effect on Brewer's sparrow of spraying big sagebrush. Journal of Range Management 28:294-297.
- Sedgwick, J.A. 2004. Habitat restoration for Gunnison and Greater sage-grouse – a literature review. USDI Bureau of Land Management, Gunnison Field Office. 65 p.
- Shaffer, T.L. 2004. A unified approach to analyzing nest success. Auk 121(2):526-540.
- Thrift, M., Magee, P.A. 2005. The effects of mowing on small mammal communities in arid sagebrush-steppe within Gunnison Basin. Thornton Final Research Report, Western State College of Colorado, Gunnison 13 p.
- Wiens, J.A., Rotenberry, J.T., Van Horne, B. 1986. A lesson in the limitations of field experiments: shrubsteppe birds and habitat alteration. Ecology 67(2):365-376.
- Winter, M., Hawks, S.E., Shaffer, J.A., Johnson, D.H. 2003. Guidelines for finding nests of passerine birds in tallgrass prairie. The Prairie Naturalist 35(3): 197-211.

Effect of Browsing Following Wildfire in the Missouri Breaks

Richard B. Keigley¹ and Kent Undlin²

ABSTRACT

This study examined the effect of browsing on skunkbush (*Rhus trilobata*) and chokecherry (*Prunus virginiana*) following two wildfires in the Missouri Breaks region of Garfield County in east-central Montana, one fire in the 2003 the other in 2006. Study objectives included: 1) Determine the potential height to which skunkbush and chokecherry can grow under local conditions, and 2) Determine if browsing will prevent young plants from attaining that potential height. Because ungulates are attracted to recently burned areas it is important to monitor browse use and regulate browsing pressure to a benign level. We documented the effect of browsing by measurement of plant height and age, browsing level, and LD Index. The most rapid growth occurred in the first two years following the fires. It is predicted that browsing will not prevent skunkbush from growing to its potential height. Three lines of evidence indicate that browsing is likely to prevent many chokecherry plants from growing to potential height. In the first few years following wildfire, the potential effect of browsing is best documented by analysis of height and age relationships and by analysis of stem growth rate. After five years browsing level and LD Index are a more-precise means of documenting the effect of browsing.

INTRODUCTION

This study examined the effect of browsing on skunkbush (*Rhus trilobata*) and chokecherry (*Prunus virginiana*) following two wildfires in the Missouri Breaks region of Garfield County in east-central Montana. The Missouri Breaks Fire complex was ignited by lightning on July 12, 2003, and burned a total of 131,000 ac before containment on July 27. The Black Pulaski Fire complex was ignited by lightning on July 16, 2006, and was contained on July 23 after burning 60,037 ac. The area provides important habitat for mule deer (*Odocoileus hemionus*) and elk (*Cervus elaphus*). The plant communities include stands of ponderosa pine (*Pinus ponderosa*) interspersed with grassland areas used for livestock grazing. The grasslands are dominated by needle- and-thread (*Stipa comata*), blue gramma (*Bouteloua gracilis*), and western wheatgrass (*Elymus smithii*). Bluebunch wheatgrass (*Elymus spicatus*) occurs in association with ponderosa pine. Skunkbush (*Rhus trilobata*) grows in association with shale outcrops; chokecherry (*Prunus virginiana*) grows on relatively moister sites. Skunkbush and chokecherry are an important

component of structural diversity and are an important source of food in the form of browse and fruit.

Because post-fire vegetation is protein rich and highly palatable (Carlson and others 1993; Cook and others 1994; Hobbs and Spowart 1984; Van Dyke and Darragh 2006ab), ungulate use is often concentrated in recently burned areas (Biondini and others 1999; Fuhlendorf and Engle 2001; Lay 1967; Pearson and others 1995; Van Dyke and Darragh 2006ab; Vermeire and others 2004). Browsing by wild and domestic ungulates can prevent young plants from attaining their potential stature (Keigley and others 2005; Keigley and Frisina 1998; Ripple and Beschta 2003; White and others 1998). If ungulate use is concentrated on Missouri Breaks and Pulaski burned areas, there is the potential that some browse species could be adversely affected.

The study objectives addressed below are: 1) Determine the potential height to which skunkbush and chokecherry can grow under local conditions, and 2) Determine if browsing will prevent young plants from attaining that potential height. This paper primarily describes the analysis of data collected in 2007; some chokecherry data collected in 2008 are also described.

METHODS

Study Sites

Study sites were established in four burned areas. The Big Coulee and Germaine Coulee areas were burned by Missouri Breaks Fire in 2003 (figure 1). The Lost Creek and Lodgepole areas were burned by the Black Pulaski Fire in 2006. The Lost Creek and Big Coulee burned areas were adjacent to one another, as were the Lodgepole and Germaine Coulee burned areas. The Lodgepole / Germaine Coulee burned areas were located approximately 20 miles southwest of the Lost Creek / Big Coulee burned areas.

In 2007 a total of 11 study sites were established in the Big Coulee (5 sites) and Germaine Coulee burned areas (6 sites). Measurements of chokecherry and skunkbush were taken at each site. The additional study site in the Germaine Coulee burned area was established to document the unusually robust growth of chokecherry that occurred in one area. A total of 17 study sites were established in the Lost Creek and Lodgepole burned areas. In the Lost Creek burned area, chokecherry and skunkbush were examined at seven sites, one of which was inside a big game enclosure, one inside a livestock enclosure, and five that were not protected from browsing. In each burned area, one of the sites exposed to browsing was located in the immediate

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Richard B. Keigley, USGS, Northern Rocky Mountain Science Center, Bozeman, MT

²Kent Undlin, Wildlife Biologist, BLM, Miles City Field Office, Miles City, MT

vicinity of the exclosures. In the Lodgepole burned area, separate big game and livestock exclosures were constructed for skunkbush and chokecherry.

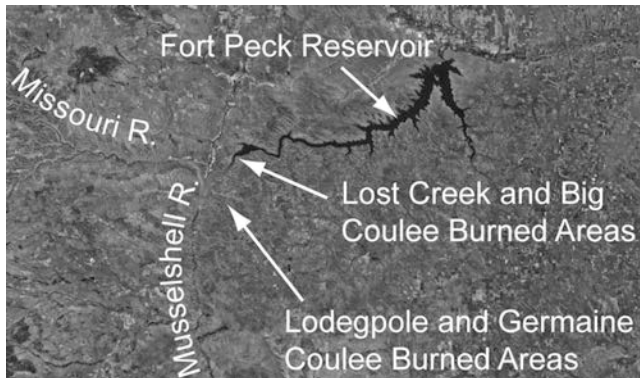


Figure 1—Location of study areas. The Lost Creek / Big Coulee burned areas are approximately 35 miles northwest of Jordan, MT.

Selection of Plants for Sampling

At each site, 20 plants of both species were selected for measurement by walking a transect line and stopping at equal intervals. The pace interval varied depending on the size of the stand. The closest live plant that was fully exposed to potential browsing was selected for measurement.

Measurements in Missouri Breaks Burned Areas

The height of live plants was measured to the base of current-year-growth (H_{BCYG}). In the case of skunkbush, the tallest stem of the shrub (as determined by height to the base of current-year-growth) was selected for measurement. If the sprouts originated from a plant that was not consumed by fire (in other words, terminal buds were present on the dead stem), the height of the parent plant was measured. The height of dead plants was measured to the tip of the stem. In cases where the parent plants were consumed by fire a separate search was conducted to complete the sample of 20 plants; at some sites 20 suitable plants were not present.

Browsing was classified into two levels by examining annual growth increments on vertically oriented stems. The length of annual growth increments was determined by inspection of bud scars. A stem was classified as intensely browsed if one or more vertically-oriented, complete annual growth increments were browsed and were dead. If no complete annual increment was browsed and dead, the stem was classified as light-to-moderately browsed. A skunkbush plant was classified as intensely browsed if any stem in the shrub was intensely browsed. A chokecherry plant was classified on the basis of its single stem. The browsing level of the site was calculated as the percent of the 20 plants that were intensely browsed.

Live / Dead Index (LD Index) was used to assess the effect of browsing on intensely browsed plants. The index was calculated as: $LD\ Index = H_{BCYG} - H_D$, where H_D is the height to the tip of the tallest dead, complete annual increment (figure 2). An LD Index of about zero would indicate that a plant was being browsed to the zone of mechanical protection (browsed, dead twig clusters often deter browsing in the interior of the plant). A highly negative LD Index would indicate that a plant was dying to ground level. A highly positive LD Index would indicate that a plant was gaining height after past intense browsing.

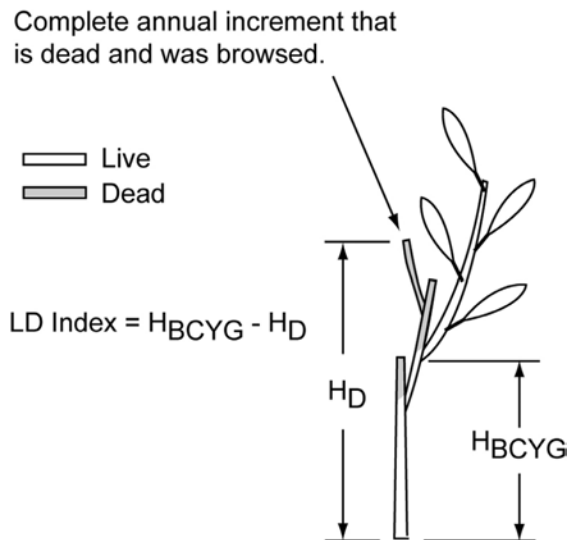
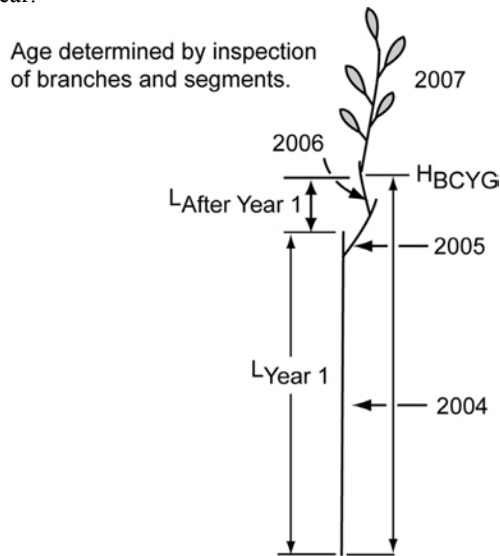


Figure 2—Measurements taken on intensely browsed plants. The measurements are used to calculate LD Index.

Over the long term, percent intensely browsed and LD Index will be used to document the effect of browsing. However, because most plants established in the Missouri Breaks burned areas were only four years old in 2007, there was concern that the plants had not been exposed to browsing long enough to develop the characteristics of intense browsing. LD Index measurements were taken in 2007 and 2008 to assess when LD Index could be used after wildfire. In the short-term, an analysis of stem growth rate was used to gain a perspective on the effect of browsing on chokecherry. The measurements on which the growth rates were based are described in figure 3.

Because skunkbush was seldom browsed, it was possible to measure the net length of stem that elongated each year after the fire. The annual increments were measured on the stem that was selected for height measurement. Annual increments were measured from the base of the increment to the point where the higher (younger) increment developed from the older stem segment. These data were used to assess the rate at which plants grew to potential stature.

For skunkbush and chokecherry the incidence of browsing during the growing season of 2007 was determined from inspection of the tallest terminal leader. For chokecherry the incidence of browsing for 2004 through 2006 was determined from inspection of the respective segments; if any of the segments that elongated in a given year was browsed, the plant was recorded as having been browsed that year.



$$GR_{Year\ 1} = L_{Year\ 1}$$

$$GR_{After\ Year\ 1} = (HBCYG - L_{Year\ 1}) / (Age - 2)$$

Figure 3—Measurements used to calculate stem growth rate. Height was measured to the base of current year growth. These data were used to calculate the number of years required for a plant to grow from its height in 2007 to 3.5 m tall.

Measurements in Black Pulaski Burned Areas

The length of 2007 current-year-growth was measured. If present, the length of the segment that elongated the year of the fire (2006) was measured. Current-year-growth was examined for evidence of browsing. The height of plants killed by the fire was measured using the protocol followed in the Missouri Breaks burned areas. The plants killed by fire were examined for morphologic evidence of browsing.

In the future, plants growing inside the big game exclosures will document the height to which plants can grow in the absence of browsing. A temporary estimate of that potential height was made from measurements of plants that were protected from browsing by topography.

RESULTS AND DISCUSSION

Skunkbush

Potential Height

The mean height of skunkbush killed by the Missouri Breaks Fire was 51 ± 3 cm ($n = 81$; $\pm$ SE); the mean height of skunkbush killed by the Black Pulaski Fire was 58 ± 1 cm ($n = 195$). These heights include the stem segment that

was current-year-growth at the time of the respective fires. The maximum height of a skunkbush killed by fire was 122 cm. In a search for taller skunkbush growing in areas protected from browsing, none were found.

Although there was morphologic evidence that skunkbush was browsed prior to the fires, the stems were not hedged. This suggests that prior to fire, browsing had a minimal influence on skunkbush height growth, and that plants likely grew to their potential stature given local site conditions. Depending on site conditions, the potential height of skunkbush likely ranges from 50 cm to somewhat over 1 m.

Effect of Browsing Following Wildfire

The percent of current-year-growth leaders that were browsed at the Lost Creek and Lodgepole burned areas was 0.7 and 1.8 percent respectively. The percent of current-year-growth leaders that were browsed at the Big Coulee and Germaine Coulee burned areas was 1.0 and 23.0 percent, respectively. At Germaine Coulee Site 5 all current-year-growth leaders were nipped. If Site 5 is excluded, 3.8 percent of leaders at Germaine Coulee were nipped. The incidence of browsing suggests that skunkbush browse use was greater at the southwestern end of the study area.

With the exception of Germaine Coulee Site 5, all Missouri Breaks burned area sites were zero percent intensely browsed. At Germaine Coulee Site 5, 65 percent of the plants were intensely browsed; the mean LD Index of those plants was -2 cm. Based on incidence of browsing and browsing level, browse use at Germaine Coulee Site 5 differed from all other sites.

Plant height and growth rate data indicate that growth to potential stature occurs within a few years. The mean height (H_{BCYG}) of skunkbush established after the Missouri Fire was 43 ± 1 cm. To compare this height to the height of plants that were killed by fire, the mean length of current-year growth (8 ± 0 cm) was added, resulting in a total height of 51 cm. After four full growing seasons, skunkbush established after the Missouri Breaks Fire attained a height similar to that of plants killed by the fire (51 cm).

The length of stem segments produced in each year following the Missouri Breaks fire indicates that growth rate declines with age (figure 4). This conclusion is based on lengths produced during the complete growing seasons of 2004 through 2007. Only 10 of the 200 sampled plants included segments that elongated in 2003. These segments completed their growth to a mean height of 12 ± 2 cm within a truncated season that began in late-July. Seventy three percent of the total stem growth in Missouri Breaks burned area occurred during the first two full growing seasons.

In the Black Pulaski burned areas, sprouts were observed in the Fall of 2006, however no segments produced in 2006 were encountered in 2007; all sampled plants consisted solely of current-year-growth. The mean height of current-year growth was 57 ± 1 cm, a height similar to the mean height of mature plants killed by the fire (58 cm).

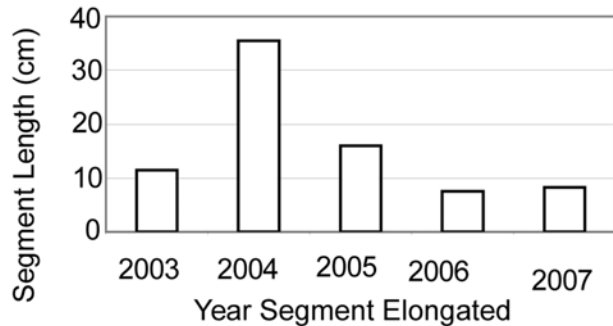


Figure 4—Much of the growth to a plant's potential height occurs during the first two years after fire. The most rapid growth occurred during the first full growing season. Growing conditions during these initial years likely plays an important role in determining a plant's ultimate stature.

In a comparison of pooled data by ANOVA, the 2007 current-year-growth of the Black Pulaski plants was significantly greater than the stem segments that elongated for the first time in 2004 in the Missouri Breaks burned areas (57 ± 1 cm and 35 ± 1 cm, respectively; $P < 0.001$). Both of these segments were in their initial growing season and could be expected to be similar in length. Figure 5 describes the mean length of the segments by year at each of the burned areas. The difference in first-season segment length may be due to differences in 2004 and 2007 precipitation.

Two Remote Automated Weather Stations documented a period of substantial precipitation in late-May of 2007; the corresponding period in 2004 was relatively dry (figure 6). In the vicinity of the Lost Creek burned area, 18 cm (7.08 in) of precipitation fell between May 20 and May 31, 2007. Over that same period in 2004, the precipitation was 4.3 cm (1.7 in). Over the same period in the vicinity of the Lodgepole burned area, 13 cm (5.13 in) of precipitation fell in 2007, compared to 5 cm (1.98 in) in 2004. The late-May precipitation in 2007 occurred immediately prior to the intense summer heat typical of the area, and may have extended the 2007 growing season.

The difference in precipitation may also explain why the 57-cm -height of the one-season old plants in the Black Pulaski burned areas was greater than the total 51-cm height of the four-season-old plants in the Missouri Breaks burned areas ($P < 0.01$; the total height of Missouri Breaks plants includes current-year-growth). Being in their initial growing season, the Black Pulaski plants were growing at a high rate; an extension of the growing season would add a correspondingly long length of stem. In contrast, most of

the Missouri Breaks plants were in their fourth growing season, and growing at a reduced rate (see figure 3). Because their rate of growth was relatively slow, an extension of the growing season would produce a correspondingly shorter length of stem. Growing conditions during the initial year of growth may be a significant factor that determines the potential height of skunkbush at a site.

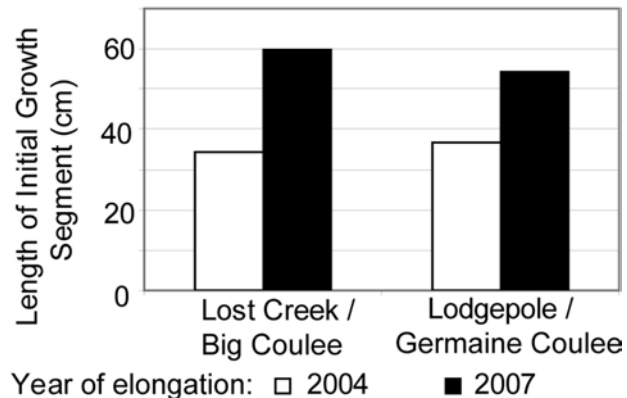


Figure 5—Skunkbush stems that elongated for the first time in 2007 grew longer than stems that elongated for the first time in 2004. A similar effect occurred with chokecherry (68 versus 56 cm). The between year difference in growth could be due to differences in precipitation described in figure 5.

Chokecherry

Potential Height

At a few locations, chokecherry plants killed by fire had grown to at least 4-m tall. The site conditions where they grew appeared to be similar to those that existed close-by where chokecherry plants were shorter. The shorter plants were hedged by browsing, while the tall plants were not. It appears that, if protected from browsing, chokecherry has the potential to grow to 3-to-4-m tall in the study area. Growth within big game exclosures will be used to reexamine this conclusion in the future.

Effect of Browsing Following Wildfire

The stems killed by fire were often hedged, suggesting that browsing had affected height growth in the past. In the Black Pulaski burned areas, 20 percent of the current-year-growth was browsed; in the Missouri Breaks burned areas 26 percent of the current-year-growth was browsed. Because measurements were taken in late-August and early-September, these data reflect browsing that occurred during the 2007 growing season. From 2004 through 2006, the incidence of browsing was 77, 86, and 80 percent, respectively. The incidence of browse use approximately triples during the late-fall and winter.

Three lines of evidence indicate that browsing will be a dominant influence on chokecherry height growth: 1)

height and age relationships, 2) browsing level and LD Index, and 3) stem growth rate.

The mean height of chokecherry killed by the Missouri Breaks Fire was 69 ± 6 cm ($n = 24$; $\pm$ SE); the mean height of chokecherry killed by the Black Pulaski Fire was 73 ± 2 cm ($n = 228$). The mean height (H_{BCYG}) of chokecherry established after the Missouri Fire (excluding Germaine Coulee Site 3b) was 71 ± 1 cm ($n = 200$). After four full growing seasons, chokecherry plants established after the Missouri Breaks Fire were approximately the same height as the plants killed by fire. The mean length of current-year-growth in the Black Pulaski burned areas was 68 ± 1 cm, a height slightly shorter than the mean height of plants killed by the Black Pulaski Fire. The similarity of heights among different age classes indicates that after the initial growing season, height growth ceases. The presence of hedging indicates that browsing is a factor.

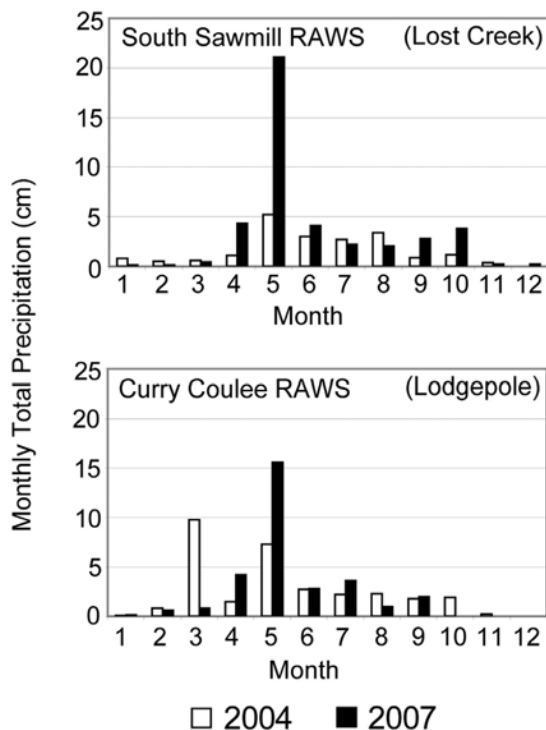


Figure 6—Precipitation in late-May of 2007 was substantially greater than that occurring in 2004. At that time of year, hot dry conditions could potentially slow growth rate. The precipitation in 2007 may have had the effect of extending the growing season. Weather conditions during the initial season of elongation could be an important determinant of potential stature.

Browsing level and LD Index is the second line of evidence suggesting that browsing will influence height growth at some sites. Data collected in 2007 and 2008 are plotted in figure 7. In 2007 three sites were 0 percent intensely browsed. Because no LD Index can be calculated for light-

to-moderately browsed plants, these sites are not depicted on the graph. The position of a point on the graph indicates the potential severity of browsing at a site. A severe browsing effect would be indicated by a clustering of points in the upper-center and upper-left of the graph. The browsing level and LD Index data collected in 2007 after the fire do not indicate a severe browsing effect.

Because most of the plants established after the Missouri Breaks Fire were four years old in 2007, it seems possible that they had not been exposed to browsing long enough to develop the characteristics of intense browsing. The data collected in 2008 corroborate this interpretation. By 2008 only a single site (GC03-2) was 0 percent intensely browsed. One site was 100 percent intensely browsed, and three sites were 95 percent intensely browsed. It is our interpretation that these changes reflect cumulative stress on plants rather than an increase in browsing pressure.

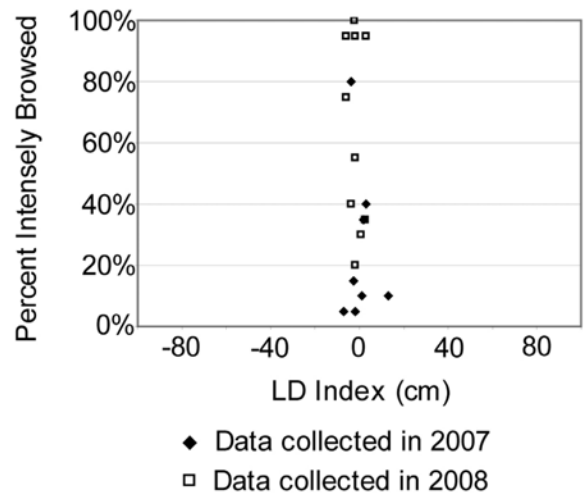


Figure 7—The relationship between percent intensely browsed and LD Index can be used to describe how the effects of browsing change over time. The earliest stage of decline would be indicated by a point at the bottom center of the graph. As a higher percentage of plants in the sample are intensely browsed, the point would move to the upper center. If browsing pressure remains high, the point would move from the upper center to the upper left. If browsing pressure diminishes, the point would move from the upper center to the upper right; as the number of light-to-moderately browsed plants increases, the point would move from the upper right to the lower right.

The analysis of stem growth rate is the third line of evidence pointing to a browsing-related effect on height growth. The goal of this analysis is to determine if a chokecherry plant will grow at a rate that will allow it to attain potential stature before it dies. The analysis was conducted separately for the Big Coulee, Germaine Coulee (excluding Site 3b), and Germaine Coulee Site 3b. Chokecherry at Germaine Coulee Site 3b was sampled because plants growing there were visibly more robust and

more lightly browsed compared to plants at other sites where site conditions were similar. For undetermined reasons, chokecherry plants in this small patch (approximately 5-by-20 m) grew to an unusual height and may demonstrate the growth potential for the area.

The mean length of the stem segments that elongated in 2004 was 56 ± 1 cm ($n = 199$) (excluding Germaine Coulee Site 3b). After that initial growing season, stem growth rate diminished with mean values ranging from 16 cm / year at Big Coulee Site 4, to -0.1 cm / year at Germaine Coulee Site 5. Thirty-six plants had a negative growth rate; after initial elongation, these plants were growing shorter. Based on each plant's growth rate, we calculated the total age at which the plant would grow from its height in 2007 to a height of 3.5 m. A frequency distribution of those ages is shown in figure 8. Plants having negative growth rates were included in the ">49 year" class.

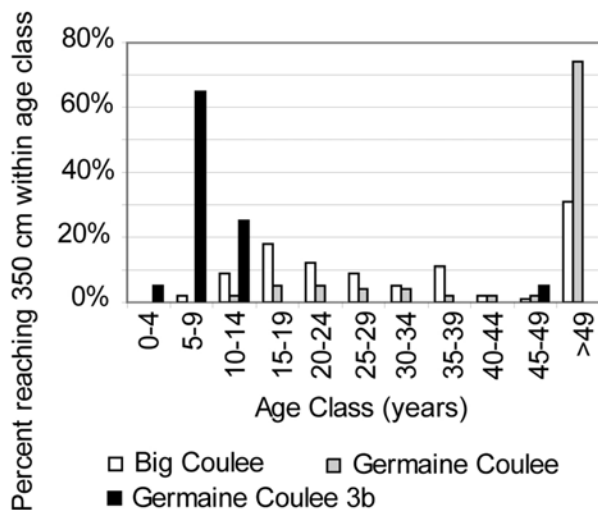


Figure 8—Frequency at which plants are predicted to grow from their height in 2007 to 2.5 m tall within a given period of years. A relatively large percentage of plants at Germaine Coulee Site 3b are predicted to grow to that height by 15 years of age; the browsing level at this site was 0 percent intensely browsed. Relatively few plants at other sites are predicted to attain 3.5 m by 15 years of age; at these areas browsing will likely prevent many plants from growing to their potential stature.

At Germaine Coulee Site 3b 95 percent of the stems are predicted to grow to 3.5 m by the time they are 15 years old. Unless conditions change, we predict that many of these plants will attain their potential stature. At Big Coulee and Germaine Coulee a respective 11 and 2 percent are predicted to grow to that height by the same age. Because growth rate diminishes with age and because browsing often shortens the lifespan of stems (as evidenced by

mortality observed in 2008), it seems likely that few Big Coulee and Germaine Coulee plants will attain potential stature.

Conditions during the first few years following wildfire are critically important to the development of the future structure of the plant community. Browsing pressure must be monitored and regulated if browsing is to be limited to levels that do not adversely affect the future structure of the plant community. During the initial years of growth, many plants may not have developed the morphologic characteristics of intense browsing; the measurement of browsing level and LD Index may not detect an adverse effect of browsing. The analysis of stem growth rate may provide an alternate means of assessing the effect of browsing during the early period. The age of young plants can easily be determined by inspection, and used to calculate growth rate. After a few years, the determination of age by inspection is problematic. At this point, browsing level and LD Index provide a better means of measuring the effect of browsing.

REFERENCES

- Biondini, M.E. et al. 1999. Bison use of fire-managed remnant prairies. *Journal of Range Management*. 52:454-461.
- Carlson, P.C. et al. 1993. Fire in Key Deer Habitat Improves Browse, Prevents Succession, and Preserves Endemic Herbs. *Journal of Wildlife Management*. 57:914-928.
- Cook, J.G., Hershey, T.J., Irwin, L.L. 1994. Vegetative response to burning on Wyoming mountain-shrub big game ranges. *Journal of Range Management*. 47:296-302.
- Fuhlendorf, S.D., Engle, D.M. 2001. Restoring heterogeneity on range lands: ecosystem management based on evolutionary grazing patterns. *Bioscience* 51:625-632
- Hobbs, N.T., Spowart, R.A. 1984. Effects of prescribed fire on nutrition of mountain sheep and mule deer during winter and spring. *Journal of Wildlife Management*. 48:551-560.
- Keigley, R.B., Frisina, M.R. 1998. Browse evaluation by analysis of growth form. *Montana Fish Wildlife and Parks*. Helena, MT. 153 pp.
- Keigley, R.B., Skinner, B., Harrell, D. 2005. Influence of ungulate browsing on skunkbush height growth on the CMR NWR. pp. 63-72 In: M.R. Frisina and S.J. Knapp (eds.) *Statewide Browse Evaluation Project Report No. 2*. Montana Fish Wildlife and Parks. Helena, MT.
- Lay, D.W. 1967. Browse palatability and the effects of prescribed burning in southern pine forests. *Journal of Forestry*. 65:826-828.
- Pearson, S.M. et al. 1995. Winter use by large ungulates following fire in Northern Yellowstone Ntl. Park. *Ecological Applications*. 5:744-755.
- Ripple, W.J., Beschta, R.L. 2003. Wolf reintroduction, predation risk, and cottonwood recovery in Yellowstone National Park. *Forest Ecology and Management*. 184:299-313
- Van De Vijver, C.A.D.M., Poot, P., Prins, H.H.T. 1999. Causes of increased nutrient concentrations in post-fire regrowth in an East African savanna. *Plant and Soil*. 214:173-185.
- Van Dyke, F., Darragh, J.A. 2006a. Short- and long-term changes in elk use and forage production in sagebrush communities following prescribed burning. *Biodiversity and Conservation*. 3:4375-4398.
- 2006b. Short- and longer-term effects of fire and herbivory on sagebrush communities in south-central Montana. *Environmental Management*. 38:365-376.
- Vermeire, L.T., et al. 2004. Patch burning effects on grazing distribution. *Journal of Range Management*. 57:248-252.
- White, C.A., Olmsted, C.E., Kay, C.E. 1998. Aspen, elk, and fire in the Rocky Mountain National Parks of North America. *Wildlife Society Bulletin*. 26:449-462.

Quaking Aspen Ecology on Forest Service Lands North of Yellowstone National Park

David S. Kimble¹, Daniel B. Tyers², and Bok F. Sowell³

ABSTRACT

Quaking aspen (*Populus tremuloides*) occupy a small area in the northern Rocky Mountains, but are highly valued as wildlife habitat. Aspen stands in and around Yellowstone National Park commonly consist of few, large, mature overstory stems and numerous root suckers that do not grow above the browsing reach (≈ 2 m) of most wild ungulates. Our primary objective was to determine if the recruitment or density of aspen stems > 2 m tall had changed from 1991 to 2006 on a portion of the Gallatin National Forest. The same aspen stands were surveyed in 1991 and 2006 in the 560 km² study area ($n = 316$). Secondary objectives were to determine if aspen density was influenced by elk (*Cervus elaphus*) browsing, conifer establishment, and cattle (*Bos spp.*) grazing. Mean recruitment stem density did not change from 1991 to 2006 ($P = 0.95$). Density of stems > 2 m declined 12 percent from 1991 to 2006 ($P = 0.04$), which indicates that recruitment stems are not being produced at a sufficient rate to replace aging overstories. Areas with the greatest elk densities had the lowest recruitment stem densities and contributed the most to the decline. Although elk browsing seemed to play the largest role, conifer establishment and cattle grazing have also negatively impacted overstory recruitment in aspen stands. Even though elk numbers on the Northern Yellowstone Winter Range have declined since wolf reintroduction, aspen recruitment has not increased at the landscape level on the Gallatin National Forest.

INTRODUCTION

Quaking aspen (*Populus tremuloides*) occupy a small area in the northern Rocky Mountains, but stand out as the only upland, deciduous tree species in much of the West (Debyle and Winokur 1985). Aspen are clonal and primarily rely on root sprouts to replace aging stems for the species to persist in the Rocky Mountains (McDonough 1985). Although aspen forests make up a small percentage of vegetative cover in the Rocky Mountains, they commonly support more species and greater numbers of wildlife than associated conifer habitats (DeByle 1985b). Wild ungulates such as moose (*Alces alces*), Rocky Mountain elk (*Cervus elaphus*), and deer (*Odocoileus spp.*) selectively browse aspen suckers in winter because they are palatable and available above the snow (Houston 1982).

Where ungulate densities are high, browsing can suppress aspen sucker growth and prevent aspen clones from replacing older overstory stems with younger stems (Debyle 1985a). Aspen coverage on the Northern Yellowstone Winter Range (NYWR) has declined from 4 to 10 percent of the landscape near the beginning of the 20th century to 1 percent presently (Wagner 2006). The NYWR is the wintering ground for Yellowstone National Park's (YNP) largest elk herd and consists of low- to mid-elevation areas in the Lamar, Yellowstone, and Gardner River drainages inside and outside YNP (Houston 1982). Although other factors may play a role, the primary cause of aspen decline on the NYWR is browsing of suckers by wintering elk (NRC 2002).

Apical dominance may impede regeneration when an aspen stand has many mature stems and auxins that suppress the growth of root sprouts are transported from the aboveground stems to the roots (Schier and others 1985). A disturbance such as fire or beaver cutting that kills the aboveground stems can end apical dominance in the clone, usually results in abundant production of root suckers in subsequent years, and may allow large numbers of suckers to eventually grow to tree size (Bartos and Mueggler 1981). Conifers are more shade tolerant than aspen and are generally associated aspen stands that have been undisturbed for many years. Eventually, conifers may invade aging aspen overstories (Jones and Schier 1985) and can suppress aspen growth (Brown and others 2006; Shepperd and others 2001).

Domestic cattle (*Bos spp.*) graze in aspen communities in some locations on the NYWR outside YNP. Domestic cattle will browse aspen and can suppress aspen growth or prevent stand regeneration under some conditions (Fitzgerald and Bailey 1984). Although elk browsing is generally accepted as the primary cause of aspen decline on the NYWR (NRC 2002), the cumulative effects of summer cattle browsing and winter wild ungulate browsing can result in decreased aspen recruitment (Kay and Bartos 2000).

St. John (1995) surveyed aspen clones or "stands" on the Gallatin National Forest (GNF) in 1990 and 1991—the largest study of its kind on the NYWR. Only 47 percent of aspen stands ($n = 342$) had recently grown any suckers above elk-browsing height and only 21 percent of stands had sufficient numbers of suckers escaping browsing for the overstories to remain stable or increase in density. Wolves (*Canis lupus*), a major predator of elk, were reintroduced

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹David S. Kimble, South Dakota Department of Game, Fish & Parks, Rapid City, SD 3305 West South Street, Rapid City, SD david.kimble@state.sd.us

²Daniel B. Tyers, USDA Forest Service, Gardiner, MT

³Bok F. Sowell, Montana State University, Bozeman, MT

into YNP in 1995 (Smith and others 2005). The NYWR elk herd decreased by ≈ 50 percent from 1995 to 2004 (NYCWWG 2005) and some researchers hypothesized that wolf predation could relieve elk browsing pressure such that aspen overstories could regenerate (Ripple and Larsen 2000). A wolf-elk-browse plant trophic cascade appears to have allowed regeneration of willow (*Salix* spp.) and cottonwood (*Populus* spp.) on small scales at certain sites in YNP (Ripple and Beschta 2003; 2004; 2006). Browsing intensity has decreased and height of the tallest cohort of aspen suckers has increased in recent years in the Lamar Valley of the NYWR (Ripple and Beschta 2007). However, it is not known if growth of the 5 tallest suckers in the stand represent a true recovery relative to pre-wolf conditions. To date, published studies have not addressed changes in broad-scale aspen overstory regeneration since wolf reintroduction.

This study repeated the 1991 aspen survey on the GNF portion of the NYWR (St. John 1995). Objectives were to determine if: 1) aspen overstory density and recruitment decreased, increased, or was stable from 1991 to 2006; 2) elk herbivory influenced aspen density; 3) conifer establishment in aspen stands influenced aspen density; and 4) cattle grazing influenced aspen density.

STUDY AREA

The study area was located in south-central Montana, primarily on the west unit of the Gardiner Ranger District (GRD), GNF and exclusive of the Absaroka-Beartooth Wilderness (figure 1). The southernmost portion of the study area abuts YNP. Mean annual precipitation in the Gardiner Basin at the southern end of the study area is 25 cm. Mean minimum temperature in January is -10°C and mean maximum temperature in July is 30°C . Elevation in the town of Gardiner is 1,618 m. Precipitation increases and temperature decreases as elevation increases in the study area. Elevation in Jardine, Montana is 1,966 m and mean annual precipitation is 45 cm (Western Regional Climate Center 2007).

Vegetation primarily consists of big sagebrush (*Artemisia tridentata* ssp.) and grassland (for example *Pseudoroegneria spicata*; *Festuca idahoensis*) at lower elevations, some quaking aspen at forest-grassland boundaries and in riparian areas, Douglas-fir (*Pseudotsuga menziesii*) at mid-elevations, and lodgepole pine (*Pinus contorta*), Engelmann spruce (*Picea engelmannii*), and sub-alpine fir (*Abies lasiocarpa*) at higher elevations (Despain and others 1986). Aspen forests range from 1,571 m to 2,605 m in elevation and occupy ≈ 745 ha of the ≈ 560 km² study area (St. John 1995).

The study area consists of three biologically dissimilar geographic regions where elk migration patterns,

population trends, and human hunting pressures differ. The study area is separated into the East River, West River, and Tom Miner Basin regions to determine if aspen density or recruitment is influenced by elk herbivory.

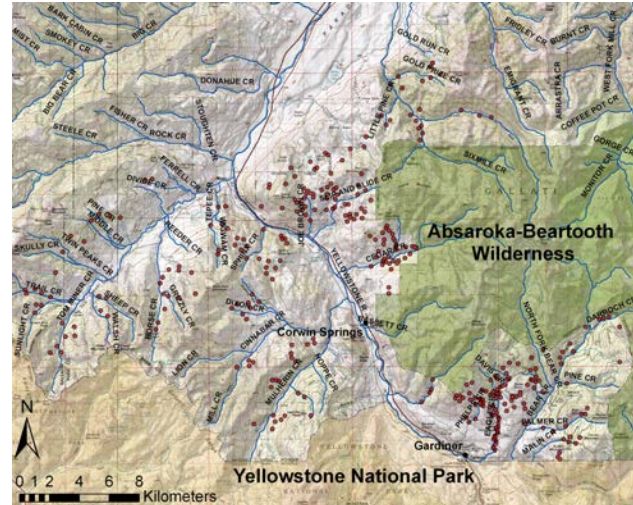


Figure 1—Sampled aspen stand locations on the study area.

The East River unit is separated from Tom Miner Basin and the West River unit by the Yellowstone River, private residences and agricultural lands in the river valley, and a major highway (figure 2). East River, at ≈ 340 km², is the largest portion of the study area. Most of the East River unit is part of the NYWR (Lemke and others 1998). From 1989 to 1999, an average of 4.8 elk/km² were annually counted by aerial census in late winter on the East River unit. About 70 percent of elk migrating north of YNP and east of the Yellowstone River have spent the majority of the winter on the state-owned Dome Mountain Wildlife Management Area since 2000. From 2000 to 2006, late winter elk density on the East River unit averaged 2.4 elk/km² (NYCWWG 2005; Tom Lemke, MFWP, unpublished data).

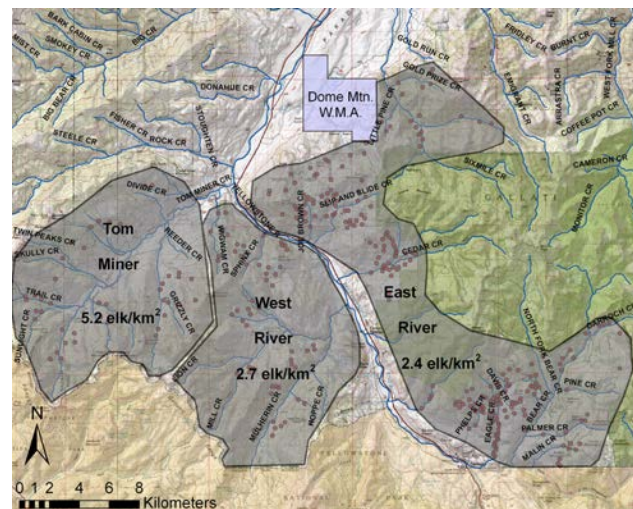


Figure 2—East River, West River, and Tom Miner subunits with different wintering elk densities.

The West River portion of the study area occupies $\approx 120 \text{ km}^2$ and is separated from Tom Miner Basin by a hydrologic divide of 2 900 m mountains in the south and private residences and agricultural land in the north (figure 2). The West River unit is also part of the NYWR (Lemke and others 1998). Average late-winter density of elk counted by aerial census was 1.9 elk/ km^2 from 1989 to 1999 and 2.7 elk/ km^2 from 2000 to 2006 (Tom Lemke, MFWP, *unpublished data*).

Tom Miner Basin occupies $\approx 100 \text{ km}^2$ (figure 2). Most of the basin bottom is private land, while Forest Service lands surround the basin at higher elevations. Tom Miner Basin is not part of the wintering range for the northern Yellowstone elk herd, but other local herds of elk winter there (Tom Lemke, MFWP, *unpublished data*). The number of elk wintering in Tom Miner Basin since 1990 has been highly variable. A high of 1 470 elk were censused in late winter in 1996 and a low of 190 elk were censused in 2004. Late winter elk density in Tom Miner Basin averaged 7.9 elk/ km^2 from 1990 to 1999 and 5.2 elk/ km^2 from 2000 to 2006 (Tom Lemke, MFWP, *unpublished data*). Despite the decline in density, average wintering elk density in Tom Miner Basin has exceeded elk density in the East and West River units since 1990.

Aspen stands located in scree communities are distributed across the study area. Aspen stands in scree communities are largely inaccessible to wild ungulates because of steep slope and rocky substrate, and can be viewed as “natural” ungulate exclosures (Larsen and Ripple 2003). Elk density in scree stands was assumed to be 0 elk/ km^2 .

METHODS

Sampling Methods

Using a combination of marked topographical maps and photographic slides taken in 1991, 342 aspen clones that St. John (1995) sampled in 1991 were revisited. Sampling methods were nearly identical to those employed by St. John (1995). After the aspen clone was relocated, the center of a 202.3 m^2 (1/20th ac) circular plot was staked within the aspen stand. The location of the plot within the stand depended upon, in ranking order: 1) where it appeared St. John's plot was located according to the photographic slide from 1991; 2) St. John's site description and site determinants such as slope, aspect, species of vegetation or number of mature aspen stems within the plot; and 3) the location most representative of the aspen stand. Each aspen stand was assigned a community type based on its associated vegetation. Aspen-conifer community types contained ≥ 10 percent canopy coverage of conifers in the stand (Mueggler 1988).

All live aspen stems within the plot were counted and categorized as sprouts ($< 1 \text{ m}$ tall), saplings (1 to 2 m tall), recruitment stems ($\geq 2 \text{ m}$ tall and $< 5 \text{ cm}$ diameter at breast height [dbh]), or mature stems ($\geq 2 \text{ m}$ tall and $> 5 \text{ cm}$ dbh). Stems were categorized by height to determine if the terminal leader had grown beyond the reach of browsing elk. Saplings (1 to 2 m) have not grown beyond the reach of browsing elk, but are not being browsed to the depth of the winter snowpack (Romme and others 1995). Stems $\geq 2 \text{ m}$ tall have grown beyond the height at which elk will browse the terminal leader (Kay 1985). Stems $\geq 2 \text{ m}$ tall and $< 5 \text{ cm}$ dbh theoretically have attained this height in recent years, while stems $> 5 \text{ cm}$ dbh represent earlier periods of recruitment in the stand (Kay 1985). Not all recruitment stems will necessarily survive as a long term component of the aspen overstory, especially where elk populations are high (Keigley and Frisina 2005). Nevertheless, this fact does not invalidate the use of recruitment stems as an index. Even if only a small proportion of recruitment stems survive to reach mature size, stands with many recruitment stems can be considered more successful at regenerating than stands with very few or no recruitment stems.

Data Analyses

Three-hundred forty-one of the 342 stands St. John sampled in 1991 were relocated and sampled in 2006. Aspen stem density data was unavailable for 1 stand St. John sampled. Twenty-four aspen stands sampled in the study area were burned or cut by beaver after 1991. Comparisons of aspen mortality and recruitment at different times and in different areas should be made among stands with similar disturbance regimes if differences are to be attributed to aspen stem senescence or ungulate impacts. Therefore, the 24 burned or beaver-cut stands were excluded from analysis. A paired *t*-test was used to compare 1991 and 2006 aspen stand density to determine if recruitment and stems $\geq 2 \text{ m}$ density had changed (McClave and Dietrich 1985). Sprout and sapling density data was not available for 1991, so all 1991 to 2006 comparisons were made only among $\geq 2 \text{ m}$ size classes.

Secondary objectives were to determine if elk herbivory, conifer establishment, or cattle grazing in aspen stands influenced aspen density. Dependent variables analyzed were: sprout density, sapling density, recruitment stem density, and stems $\geq 2 \text{ m}$ density. Paired *t*-tests were used to compare 1991 and 2006 recruitment and stems $\geq 2 \text{ m}$ densities within cattle use categories. *F*-tests from the general linear model of analysis of variance were used to compare 2006 aspen densities in the elk use and conifer categories. The $y^{0.25}$ power transformation was used to correct nonnormality in the random errors of all dependent variables (Kuehl 1994). Differences were considered significant at $P \leq 0.10$. The SAS[®] 9.1 software program was used for all statistical analyses.

RESULTS AND DISCUSSION

Aspen Sustainability

Recruitment stem density did not change across the study area from 1991 to 2006. During this time, density of stems ≥ 2 m tall declined 12 percent (table 1). Mortality of all stems that have grown above browsing height is greater than recruitment of new stems to this height in the average aspen stand in the study area. About 20 percent of aspen overstory stems on the GNF portion of the NYWR were established before 1900 (Larsen and Ripple 2003), are nearing the end of their average maximum lifespan of 120 years, and can be expected to deteriorate rapidly (Mueggler 1989). As aging overstory stems die at a faster rate, they must also be replaced with recruitment stems at a faster rate if aspen stands in this study area are to remain stable or increase in density.

Table 1—Mean recruitment and stem ≥ 2 m densities for undisturbed aspen stands on the entire study area sampled in 1991 and 2006 on the Gallatin National Forest.

	1991 (<i>n</i> = 316)	2006 (<i>n</i> = 316)	Percent Change	<i>P</i>
Recruitment				
Stems ≥ 2 m and < 5 cm dbh / ha	467.2	463.2	--	0.95
Stems ≥ 2 m / ha (total overstory)	1 257.6	1 106.3	-12	0.04

^a*P*-values based on paired *t*-tests.

Aspen canopy coverage in the Greater Yellowstone Ecosystem (GYE) declined 10 percent from 1956 to 2001 (Brown and others 2006). Aspen overstory density in our study area has declined a similar amount in less time (12 percent in 15 years). Aspen coverage is declining 0.2 percent annually on average in the GYE (Brown and others 2006), but aspen density in our study area is declining 0.8 percent annually. Remotely-sensed aspen canopy coverage (Brown and others 2006) and aspen density perhaps cannot be directly compared. As aspen stems senesce, it is likely that aspen stand density will decline before stand peripheries are significantly reduced. However, aspen canopy coverage has declined 0.6 percent annually from 1958 to 1995 on the GNF portion of the NYWR within our general study area (Larsen and Ripple 2005). Our 0.8 percent annual loss in overstory density is comparable to Larsen and Ripple's (2005) 0.6 percent annual canopy loss. Therefore, the 4-fold difference in rate of canopy loss on the GYE (Brown and others 2006) and overstory density loss on our study area is still notable. Most aspen stands in the GYE are not located on heavily used ungulate winter ranges and are not in an accelerated state of decline (Brown and others 2006). Compared to the average aspen stand in

surrounding areas in Montana and Wyoming (Brown and others 2006), our 12 percent decline in density in 15 years is severe.

Aspen overstory density has declined 2.5 percent per year on the Gros Ventre elk winter range in western Wyoming (Hart and Hart 2001). Stems ≥ 2 m density declined less than 1 percent annually from 1991 to 2006 on our study area. Differences are likely attributable to an older overstory (Krebill 1972) and higher elk density (≈ 10 elk/km²) (Kilpatrick and Abendroth 2001) on the Gros Ventre winter range than in our study area.

Elk Browsing

Because elk browsing is the major influence on aspen stand regeneration in this area (NRC 2002), we can better understand aspen sustainability by separating the study area into 3 distinct regions where elk migration patterns, elk population trends, and hunting pressures on elk differ: East River, West River, and Tom Miner. In addition, aspen stands in scree communities largely unimpacted by elk.

Current stem densities were compared in each of the 4 elk use categories (table 2). Sprout (< 1 m tall) density is lowest in scree stands, probably because of the poor growing conditions on the dry, rocky substrate in scree communities. The highest recruitment stem and stem ≥ 2 m density is in scree stands (table 2). Absence of elk browsing enhances the ability of aspen stems in scree communities to grow above 2 m (Larsen and Ripple 2005). The greater density of overstory stems in scree stands is even more striking considering the low numbers of root sprouts produced in scree stands. A large fraction of root sprouts produced in scree stands apparently survive to grow above 2 m.

Sprout densities do not differ among Tom Miner Basin, the West River unit, and the East River unit. Sapling, recruitment stem, and stem ≥ 2 m densities are greater in the East River unit than Tom Miner Basin or the West River unit (table 2). Because sprout densities do not differ among Tom Miner, West River, and East River aspen stands, the ability of stands in the 3 areas to produce root suckers appears similar. Mueggler (1989) estimated that aspen stands with > 2 500 suckers (stems < 1.4 m tall) / ha had the potential to successfully replace the overstory.

Average sprout densities for aspen stands in each of the 3 areas exceed this number. However, densities of stems in size classes > 1 m are lower in the West River unit and Tom Miner Basin than in the East River unit. Elk browsing prevents root suckers from growing above 1 m elsewhere on the NYWR (Romme and others 1995). Elk browsing is probably more effectively preventing sprouts from increasing in height in the West River unit and Tom Miner Basin than in the East River unit. Elk impacts on recruitment are likely more severe in Tom Miner Basin simply because elk density is higher (White and others 2003).

Table 2—Mean sprout, sapling, recruitment stem, and stem ≥ 2 m densities in 2006 in scree communities, East River unit, West River unit, and Tom Miner Basin, Gallatin National Forest.

	Scree 0 elk / km ² (n = 23)	East River 2.4 elk / km ² (n = 203)	West River 2.7 elk / km ² (n = 50)	Tom Miner 5.2 elk / km ² (n = 40)
Sprouts < 1 m / ha	1 315.0 ^a	4 181.8 ^b	4 475.6 ^b	3 974.7 ^b
Saplings 1-2 m / ha	855.2 ^a	1 946.7 ^b	918.2 ^a	1 287.4 ^a
Recruitment Stems ≥ 2 m and < 5 cm dbh / ha	876.7 ^a	530.2 ^b	182.9 ^c	236.0 ^c
Stems ≥ 2 m / ha (total overstory)	1 566.4 ^a	1 254.5 ^b	563.4 ^c	768.5 ^c

^aDifferent superscripts represent means that differ ($P \leq 0.10$) based on ANOVA contrasts.^bComparisons made in rows.

The reasons for different densities of stems > 1 m in the East and West River unit may be more complex than simple differences in elk densities (2.4 elk/km² vs 2.7 elk/km²). The elk populations' different risks of predation in the two areas could account for the different stem densities. Elk have been hunted in winter in the East and West River units since 1976 (NYCWWG 2005). However, public hunting access to Forest Service land in the West River unit is limited because of interspersed private landholdings. The greater hunting disturbance may have caused elk to browse aspen stands less efficiently (Laundre and others 2001; White and others 2003) in the East River unit than the West River unit. Also, the population trend of the elk herd could be more important in preventing sapling and recruitment stem growth than elk densities. The decreasing elk numbers on the East River unit may have created conditions more favorable for the escape of new suckers above browsing height than the increasing elk numbers on the West River unit.

Conifer Establishment

Conifers often colonize aspen stands that have not recently burned and could reduce aspen regeneration. Aspen stands were divided into 2 groups with and without ≥ 10 percent conifer coverage in the stand. Sprout density does not differ in aspen stands with and without ≥ 10 percent conifer canopy coverage. Density of saplings, recruitment stems, and stems ≥ 2 m is approximately 2 times greater in stands with < 10 percent conifer canopy coverage than in stands with ≥ 10 percent conifer canopy coverage (table 3). Browsing seems to more effectively prevent sucker growth above 1 m in stands with conifers than in stands without conifers. Mean sprout density in aspen stands with ≥ 10 percent conifer coverage is still sufficiently high to replace the overstory (Mueggler 1989) and does not differ from mean sprout density in stands with < 10 percent conifer coverage. Because extensive root suckering occurs even in late-seral aspen stands that have been colonized by conifers,

the theory of apical dominance preventing the initiation of root suckers (Schier and others 1985) does not seem to apply to most stands in this study area. In Utah, aspen stands inside ungulate exclosures regenerate overstory stems even if they are colonized by conifers (Kay and Bartos 2000).

Aspen grow more slowly in the shade of conifers (Shepperd and others 2001), which probably plays a larger role in the failure of conifer-colonized stands of this study area to produce recruitment stems than decreased numbers of suckers. Where sucker growth is slow, less frequent browsing is required to prevent growth above 2 m than where growth is faster. Also, aspen stands colonized by conifers could be more attractive to wintering elk than stands where conifers have not established because they simultaneously provide forage, thermal cover, hiding cover, and lower snow depths. The presence of a conifer canopy appears to exacerbate the effects of browsing ungulates.

Table 3—Mean sprout, sapling, recruitment stem, and stem ≥ 2 m densities in stands with and without ≥ 10 percent conifer coverage on the Gallatin National Forest.

	Conifers $\geq$ 10% Canopy Coverage (n = 173)	Conifers < 10% Canopy Coverage (n = 143)	P ^a
Sprouts < 1 m / ha	3 934.8	4 064.3	0.44
Saplings 1-2 m / ha	1 142.7	2 199.8	0.004
Recruitment Stems ≥ 2 m and < 5 cm dbh / ha	292.5	669.8	0.0001
Stems ≥ 2 m / ha (total overstory)	731.0	1 560.4	0.0001

^aP-values based on the F-test from ANOVA.

Table 4—Mean density of recruitment stems and stems ≥ 2 m in 1991 and 2006 on the Gallatin National Forest where cattle grazed from 1991 to 2006, grazing ended between 1991 and 2006, or cattle did not graze from 1991 to 2006.

		1991	2006	Percent Change	<i>P</i> ^a
Grazed 1991 to 2006 (<i>n</i> = 104)	Recruitment stems / ha	384.4	279.4	--	0.31
	Stems ≥ 2 m / ha (total overstory)	1 055.4	776.5	-26	0.003
Grazed 1991, not 2006 (<i>n</i> = 61)	Recruitment stems / ha	151.5	520.9	+244	0.01
	Stems ≥ 2 m / ha (total overstory)	836.9	1026.5	--	0.15
Not Grazed 1991 to 2006 (<i>n</i> = 128)	Recruitment stems / ha	596.5	510.8	--	0.48
	Stems ≥ 2 m / ha (total overstory)	1 576.8	1 329.7	-16	0.08

^a*P*-values based on paired *t*-tests.

Cattle Grazing

Cattle graze in summer and early fall in some aspen stands on the study area. We compared aspen density in stands where cattle grazed from 1991 to 2006, cattle grazing ended between 1991 and 2006, and cattle did not graze from 1991 to 2006. Recruitment stem density did not change in stands where cattle grazed from 1991 to 2006 or in stands not grazed from 1991 to 2006. However, recruitment stem density increased more than 3-fold in stands where cattle grazing ended between 1991 and 2006 (table 4).

Even though stocking densities on Forest Service allotments are low (3 to 7 ha/animal unit month), the removal of cattle seems to have allowed more suckers to grow above 2 m. Because many aspen stands in the study area are heavily browsed by elk (St. John 1995), even light stocking of cattle seems to negatively impact recruitment. However, increases in recruitment did not occur in every stand where cattle were removed. In fact, 54 percent of stands where cattle were removed still grew no recruitment stems. Since conifer establishment and elk browsing are also associated with the failure to produce recruitment stems in this area, it is likely these factors must be limited if recruitment stem production is to increase after the removal of cattle.

CONCLUSIONS

Overstory aspen stems are dying more rapidly than they are being replaced in the average stand on the study area. Although NYWR elk numbers have declined since 1991, aspen recruitment has not increased and current overstory densities are apparently not sustainable on a landscape scale.

After their reintroduction in 1995, researchers hypothesized that wolves could reduce the NYWR elk population such that aspen regeneration would increase (Ripple and Larsen 2000). However, an overall increase in aspen overstory replacement has not occurred over our 560 km² study area since 1991. At this point, it does not appear that a wolf-elk-aspen trophic

cascade has allowed aspen recruitment to increase at the landscape level on the GNF north of YNP.

Aspen recruitment may have failed to increase in response to reduced wintering elk numbers in the East River unit for two likely reasons: First, more time could be required for aspen suckers “released” from browsing pressure to grow above browsing height. The number of wintering elk in the East River unit did not decline appreciably until 2000. If aspen sucker heights had been suppressed to 20 to 30 cm as in YNP (Romme and others 1995), it could take 6 years for completely unbrowsed suckers to reach 2 m in height (McColley 2007). Second, elk numbers could still be too high for aspen regeneration to increase. Because aspen are highly preferred forage in winter and occupy such a small proportion of the landscape (Houston 1982), low densities of elk may continue to suppress recruitment.

Conifer establishment and cattle grazing in aspen stands also browsed by elk appear to more effectively suppress aspen recruitment than elk browsing alone. Nonetheless, elk browsing seems to be the primary cause of aspen decline, as evidenced by high aspen recruitment and overstory densities in scree communities.

MANAGEMENT IMPLICATIONS

Management may need to change if present aspen overstory densities are to be maintained on this portion of the GNF. Increased hunter harvest of elk, removal of conifers, or removal of cattle might accomplish this goal. Gardiner Late Hunt elk harvests have been drastically reduced in recent years in Montana Hunting District 313 (East and West River units, roughly) in response to declining numbers of elk migrating north from YNP. However, wintering elk numbers in the West River unit have increased since the 1990s. Regulations could be changed so that elk harvest west of the Yellowstone River could increase, while harvest east of the Yellowstone River could remain low. Cutting conifers out of

the canopy of aspen stands could increase aspen recruitment by increasing solar radiation and growth rates of aspen suckers. It is not necessary to cut or burn the aspen canopy to promote suckering in most stands. Cutting the conifers but leaving the aspen overstory is a “safer” strategy for stands that could receive high elk use. Removal of cattle from Forest Service allotments would promote recruitment in some areas. However, the interaction of conifers slowing sucker growth and continued elk browsing will probably prevent a detectable response in some aspen stands.

REFERENCES

- Bartos, D.L., Mueggler, W.F. 1981. Early succession in aspen communities following fire in western Wyoming. *Journal of Range Management* 34:315-318.
- Brown, K., Hansen, A.J., Keane, R.E., Graumlich, L.J. 2006. Complex interactions shaping aspen dynamics in the Greater Yellowstone Ecosystem. *Landscape Ecology* 21: 933-951.
- Debyle, N.V. 1985a. Animal Impacts. *In*: Debyle, N. eds. *Aspen: ecology and management in the western United States*. GTR-RM-119., Fort Collins, CO: Rocky Mountain Forest and Range Experiment Station. 283p.
- Debyle, N.V. 1985b. Wildlife. *In*: Debyle, N. eds. *Aspen: ecology and management in the western United States*. GTR-RM-119, Fort Collins, CO: Rocky Mountain Forest and Range Experiment Station. 283p.
- Debyle, N.V., Winokur, R.P. 1985. Introduction. *In*: Debyle, N. eds. *Aspen: ecology and management in the western United States*. GTR-RM-119. Fort Collins, CO: Rocky Mountain Forest and Range Experiment Station. 283p.
- Despain, D.G., Houston, D.B., Meagher, M., Schullery, P. 1986. Wildlife in transition—man and nature on Yellowstone’s northern range. Boulder, CO: Roberts Rinehart, Inc. 142p.
- Fitzgerald, R.D., Bailey A.W. 1984. Control of aspen regrowth by grazing with cattle. *Journal of Range Management* 37:156-158.
- Hart, J.H., Hart, D.L. 2001. Interaction among cervids, fungi, and aspen in northwest Wyoming. *In*: Shepperd, W.D. et al. eds. *Sustaining aspen in western landscapes: symposium proceedings*; 13-15 June 2000; Grand Junction, CO. Proceedings RMRS-P-18. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station. 460p.
- Houston, D.B. 1982. *The Northern Yellowstone Elk; Ecology and Management*. New York, NY: Macmillan. 474p.
- Jones, J.R., Schier, G.A. 1985. Growth. *In*: Debyle, N. eds. *Aspen: ecology and management in the western United States*. GTR-RM-119., Fort Collins, CO: Rocky Mountain Forest and Range Experiment Station. 283p.
- Kay, C.E. 1985. Aspen reproduction in the Yellowstone Park-Jackson Hole area and its relationship to the natural regulation of ungulates. *In*: Workman, G. ed. *Western elk management: a symposium*. Utah State University. Logan, UT. 213p.
- Kay, C.E., Bartos, D.L. 2000. Ungulate herbivory on Utah aspen: assessment long exclosures. *Journal of Range Management* 53:145-153.
- Keigley, R.B., Frisina, M.R. 2005. Aspen height, stem-girth and survivorship in area of high ungulate use. *In*: Frisina, M and S Knapp eds. *Statewide browse evaluation project report number two*. Montana Fish, Wildlife, and Parks. 110p.
- Kilpatrick, S., Abendroth, D. 2001. Aspen response to prescribed fire and wild ungulate herbivory. *In*: Shepperd, W. et al. eds. *Sustaining aspen in western landscapes: 13-15 June 2000; Grand Junction, CO. Proceedings RMRS-P-18*. Rocky Mountain Research Station. 460p.
- Krebill, R.G. 1972. Mortality of aspen on the Gros Ventre elk winter range. Research Paper INT-129. Ogden, UT: USDA, Forest Service, Intermountain Forest and Range Experiment Station. 16p.
- Kuehl, R.O. 1994. *Statistical principles of research design and analysis*. Belmont, CA: Duxbury Press. 686p.
- Larsen, E.J., Ripple, W.J. 2003. Aspen age structure in the northern Yellowstone ecosystem. *Forest Ecology and Management* 179:469-482.
- Larsen, E.J., Ripple, W.J. 2005. Aspen stand conditions on elk winter ranges in the northern Yellowstone ecosystem. *Natural Areas Journal* 25:326-338.
- Laundre, J.W., Hernandez, L., Altendorf, K.B. 2001. Wolves, elk, and bison: reestablishing the “landscape of fear” in Yellowstone National Park, Canadian Journal of Zoology 79:1401-1409.
- Lemke, T.O., Mack, J.A., Houston, D.B. 1998. Winter range expansion by northern Yellowstone elk herd. *Intermountain Journal of Sciences* 4:1-9.
- McClave, J.T., Dietrich II, F.H. 1985. *Statistics; Third Edition*. San Francisco, CA: Dellen. 823p.
- McColley, S.D. 2007. Restoring riparian aspen stands with beaver on the Northern Yellowstone Winter Range [thesis]. Bozeman, MT: Montana State University. 67p.
- McDonough, W.T. 1985. Sexual reproduction, seeds, and seedlings. *In*: Debyle, N. and R. Winokur eds. *Aspen: ecology and management in the western United States*. Gen.Tech.Rep.RM-119., Fort Collins, CO: USDA, Rocky Mountain Forest and Range Experiment Station. 283p.
- Mueggler, W.F. 1988. Aspen community types of the Intermountain Region. General Technical Report INT-250. Ogden, UT: USDA, Forest Service, Intermountain Forest and Range Research Station 135 p.
- Mueggler, W.F. 1989. Age distribution and reproduction of intermountain aspen stands. *Western Journal of Applied Forestry* 4:41-45.
- NYCWWG (Northern Yellowstone Cooperative Wildlife Working Group). 2005. 2005 Annual Report.
- NRC (National Research Council). 2002. *Ecological dynamics on Yellowstone’s northern range*. Wash., D.C.: National Academies Press. 180p.
- Ripple, W.J., Beschta, R.L. 2003. Wolf reintroduction, predation risk, and cottonwood recovery in Yellowstone National Park. *Forest Ecology and Management* 184: 299-313.
- Ripple, W.J., Beschta, R.L. 2004. Wolves, elk, willows, and trophic cascades in the upper Gallatin Range of southwestern Montana, USA. *Forest Ecology and Management* 200: 161-181.
- Ripple, W.J., Beschta, R.L. 2006. Linking wolves to willows via risk-sensitive foraging by ungulates in the northern Yellowstone ecosystem. *Forest Ecology and Management* 230:96-106.
- Ripple, W.J., Beschta, R.L. 2007. Restoring Yellowstone’s aspen with wolves. *Biological Conservation* 138:514-519.
- Ripple, W.J., Larsen, E.J. 2000. Historic aspen recruitment, elk, and wolves northern Yellowstone National Park. *Biological Conservation* 95: 361-370.
- Romme, W.H., Turner, M.G., Wallace, L.L., Walker, J.S. 1995. Aspen, elk, and fire in northern Yellowstone National Park. *Ecology* 76: 2097-2106.
- Schier, G.A., Jones, J.R., Winokur, R.P. 1985. Vegetative Regeneration. *In*: Debyle, N. and R. Winokur eds. *Aspen: ecology and management in the western United States*. Gen Tech Rep RM-119., Fort Collins, CO: Forest Service, Rocky Mountain Forest and Range Experiment Station. 283p.
- Shepperd, W.D., Bartos, D.L., Mata, S.A. 2001. Above- and below-ground effects of aspen clonal regeneration and succession to conifers. *Canadian Journal of Forest Research* 31: 739-745.
- Smith, D.W., Stahler, D.R., Guernsey, D.S. 2005. *Yellowstone Wolf Project: Annual Report, 2004*. Yellowstone Center for Resources YCR-2005-02., Yellowstone National Park, Wyoming: USDI, National Park Service, Yellowstone Center for Resources. 18p.
- St. John, R.A. 1995. Aspen stand recruitment and ungulate impacts: Gardiner Ranger District [thesis]. Missoula: The University of Montana. 92p.
- Wagner, F.H. 2006. *Yellowstone’s destabilized ecosystem: elk effects, science, and policy conflict*. New York, Oxford University Press. 371p.
- Western Regional Climate Center. 2007. Gardiner, MT and Jardine, MT climate summaries. Available at: <http://www.wrcc.dri.edu/summary/climsmmt.html>. Accessed January 17, 2007.
- White, C.A., Feller, M.C., Bayley, S. 2003. Predation risk and the functional response of elk-aspen herbivory. *Forest Ecology and Management* 181: 77-97.

Restoring Aspen Riparian Stands With Beaver on the Northern Yellowstone Winter Range

Samuel D. McColley¹, Dan B. Tyers², and Bok F. Sowell³

ABSTRACT

Aspen (Populus tremuloides) on the Gardiner Ranger District, Gallatin National Forest, have declined over the last half-century. In an attempt to reverse this trend, beaver (Castor canadensis) were reintroduced in Eagle Creek in 1991. In 2005, we assessed the long-term effects of beaver on aspen stands and the associated riparian area in the Eagle Creek drainage. Aspen recovery was estimated by comparing vegetative changes among control sites with <10 percent beaver use (n = 5), active beaver sites (n = 6), sites abandoned for 1 to 3 years (n = 7), sites abandoned for 4 to 6 years (n = 4), and sites abandoned for 7 to 11 years (n = 5). Aspen stem densities in active sites and sites abandoned by beaver for 1 to 3 years were similar (2.6/m²) and greater (P = 0.01) than the remaining sites. Sprout and sapling densities were greater (P = 0.01) on active and sites abandoned for 1 to 3 years compared to the other sites. Aspen suckers were not able to grow taller than 2m on sites without beaver activity for 4 to 1 years, which prevented aspen recovery. Beaver activity stimulated the growth of aspen sprouts and saplings, but ungulate herbivory prevented successful aspen recovery in Eagle Creek.

INTRODUCTION

Aspen (*Populus tremuloides*) is one of the few deciduous tree species in the northern Rocky Mountains and has occupied this area for thousands of years (Jones and DeByle 1985). Aspen communities are a small portion of the vegetative cover, but sustain abundant and diverse wildlife species (DeByle 1985). The canopy cover of aspen has declined over the last century on the northern Yellowstone winter range (NYWR) (Houston 1982; Kay 1990). Elk (*Cervus elaphus*) herbivory has reputedly been the most significant contributor to this decline (Kay 1990; St. John 1995; NRC 2002; Wagner 2006). One-third of the NYWR is located on the Gardiner Ranger District, Gallatin National Forest (GNF). The GNF is directed in the Forest Plan to provide wildlife habitat by maintaining and enhancing aspen stands (GNF 1987). In an effort to comply with this mandate, beaver (*Castor canadensis*) were reintroduced into Eagle Creek, Gardiner Range District, in 1991 (D. Tyers pers. comm. 2005).

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Samuel D. McColley, Department of Animal and Range Sciences, Montana State University-Bozeman, MT

²Dan B. Tyers, Wildlife Biologist, Gallatin National Forest, Gardiner Ranger District, Gardiner, MT

³Bok F. Sowell Department of Animal and Range Sciences, Montana State University-Bozeman, MT

Beaver create disturbances in aspen stands by cutting down the overstory. This species usually responds by increasing the production of vegetative stems, or ramets. Beaver utilize aspen as a food source and as construction material for dams and lodges. Dams impound water which promotes the growth of woody plants, including aspen. Beaver often deplete their food resources and consequently disperse to a different segment of the stream. In the absence of beaver, ramets may be able to grow into mature stems. Recent surveys (Smith 2005 unpub) indicate beaver are virtually nonexistent in aspen riparian communities across the NYWR, although historically they were abundant (Warren 1926). Currently, Eagle Creek is the only area on the NYWR that has the once common ecological assemblage of beaver, aspen, and elk. This unique area presents an opportunity to study how aspen stands respond to beaver disturbance in riparian areas with high ungulate densities.

Warren (1926) and Jonas (1955) speculated that aspen stands might recover 20 to 30 years after beaver disturbance on the NYWR. Although aspen stands have only been free of beaver activities for no more than 11 years in Eagle Creek, findings from this project will help managers predict the long-term effects of beaver on aspen under current condition on the NYWR. These insights, in turn, may help explain the overarching ecological processes at work on this important ungulate winter range.

In this study we evaluated the condition of aspen riparian stands in Eagle Creek 15 years following beaver reintroduction. We investigated if beaver were able to stimulate vegetative growth in these stands through overstory cutting and the creation of water impoundments. In addition, we investigated the potential for new sprouts to survive herbivory long enough to grow into the overstory given current ungulate densities. Our objective was to determine if aspen were able to produce enough recruitment stems to sustain the stand after disturbance and if ungulate herbivory was influencing aspen recovery after beaver disturbance.

STUDY SITE

Eagle Creek is located 4 km northeast of Gardiner, Montana. It is a second order stream, with a 7 percent overall slope. The Rosgen classification for this stream is an A4. This classification indicates the stream has a moderate slope, moderate sinuosity, and gravelly substrate (Rosgen and

Silvey 1996). Davis Creek flows into Eagle Creek at the northern end of the drainage and beaver activity on this tributary was included in this study. The elevation of the 5-km stream reach we studied ranges from 1800m to 2100m. Aspen, willow, sedges and grasses are the main vegetative components with inter-dispersed Engelmann spruce (*Picea engelmannii*) within the riparian area of Eagle Creek. The dominant willow species in the Eagle Creek drainage in order of abundance are, Bebb willow (*Salix bebbiana*), Geyer's willow (*Salix geyeriana*), and sandbar willow (*Salix exigua*). Other common woody species include: mountain alder (*Alnus incana*), chokecherry (*Prunus virginiana*), and serviceberry (*Amelanchier alnifolia*). Upland-sagebrush grasslands surround the perimeter of the riparian area and are dominated by bluebunch wheatgrass (*Agropyron spicatum*), Idaho fescue (*Festuca idahoensis*), Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) and mountain big sagebrush (*Artemisia tridentata* spp. *vaseyana*).

METHODS

Beaver activity throughout Eagle Creek for the last 15 years was identified by using the USFS historical inventory. Because beaver tend to move after a site is depleted of woody vegetation, we were able to use occupancy data from the USFS inventory to determine how long it takes aspen and other woody plants to recover from beaver disturbance. To facilitate analysis, sites were grouped into three categories: active, abandoned, and control sites. Five control sites (<10 percent beaver use on woody plants) were identified throughout this stream reach. Twenty-two other sites were identified as having past or present beaver activity. Active sites had beaver present within the last 1 year. Inactive sites were identified as having past beaver activity, but the sites have been abandoned for more than 1 year. Six active and 16 abandoned sites were identified. I further classified the abandoned sites into 3 categories: 1 to 3 ($n = 7$), 4 to 6 ($n = 4$) and 7 to 11 ($n = 5$) years since abandonment. The division into 3-year increments was based on a previous study that reported major changes in woody plants 3 years after beaver abandonment (Donkor and Fryell 1999).

In the summer of 2005, woody vegetation was sampled at each site by establishing 30-1m² plots on multiple line transects within aspen stands and were distributed so they proportionately represented each stand. All woody vegetation in each plot was identified to the species level. Aspen stems were classified into different basal diameter classes based on their diameter at 30cm (Johnson and Naiman 1990). Diameter was measured at 30cm rather than the conventional "diameter at breast height" (DBH defined as 1.37m) because the majority of aspen stems in disturbed

sites were ramets < 2m in height. The diameter classes were: sprout 0 to 1cm, sapling 1 to 2cm, pole 2 to 12.5cm, and mature >12.5 (Hann and Jensen 1987). Each diameter class represents a different age class of aspen stems. The composition of diameter classes at each site gave a distribution of age classes. Aspen stem density and stem density for each diameter class was derived at each site from this information. Ungulate use was also recorded for aspen ramets in each 30 to 1m² plot during the sampling of the line transects. Ungulate use was defined as the percentage of branches on a ramet removed by ungulates within the current year.

In the summer of 2006, multiple belt transects were established at each site with a sample area equaling 60m² within each stand. The belt transects were laid out similarly to the line transects used in 2005 and followed the procedures used by Kay (1990). We used these belt transects primarily to assess the height of aspen stems to determine if ramets were recovering and growing into the overstory after beaver disturbance. For ramets to grow into the overstory they had to escape the effects of ungulate browsing. Elk are able to browse above 2m (Keigley and others 2002). However, the browsing zone of most ungulates is <2m in height, and if the terminal leader exceeds this height, the stem may have escaped most effects of browsing (Kay 1990).

The line transects sampled in 2005 did not take into consideration the height of aspen stems, therefore this method was added. I categorized all aspen stems on the belt transects into height/DBH size-classes. The height/DBH size-class categories were sprouts, saplings, recruitment poles, non-recruitment poles and mature trees. Sprouts were identified as stem <1m in height whereas saplings are between 1 and 2m in height (Hann and Jensen 1987). Recruitment poles were identified as aspen stems >2m in height and have a DBH less than 5cm (Kay 1990; Bartos and others 1991). Non-recruitment poles were identified as aspen stems >2m in height with a DBH between 6 to 9cm. If a stem has reached the non-recruitment pole criteria it is likely that it is in the overstory and is not consider a recruitment stem. Mature trees were >2m in height with a DBH >9cm. Recruitment refers to the process of younger stems replacing mature stems. We examined aspen height/DBH size-classes to determine if stems were able to recover after beaver disturbance. If aspen stems reached the recruitment pole criteria (>2m in height and <5cm DBH) then we determined that recruitment was occurring. Stegeman's (1954) analysis of aspen stem growth suggests that it would take <6 years to achieve the recruitment pole criteria.

ANOVA for unequal sample sizes was used to test for differences in aspen density, 30cm diameter class density, height/DBH size-class density, and ungulate use among control sites, active sites, sites inactive for 1 to 3 years, sites inactive for 4 to 6 years, and sites inactive for 7 to 11 years (Devore and Peck 2001, SAS 9.1). A Duncan multiple comparison procedure was completed for each variable to assess how each treatment differed from one another. Each variable was tested for normality using the Shapiro-Wilk procedure (D'Agostino and others 1990). All results were considered significant at a 0.05 α level.

RESULTS

Active sites and sites inactive for 1 to 3 years had the highest aspen stem densities when compared to all other sites, with 2.7 and 2.6 stems/m² respectively (table 1). The 30cm diameter class densities differed between the treatments in the sprout, sapling and mature categories (table 1). Sprout densities were highest in active sites and sites abandoned for 1 to 3 years. Similarly, the sapling densities were greatest for the active sites and sites abandoned for 1 to 3 years. The 30cm diameter class densities did not differ for the pole category across all sites. Mature stem density was lowest for the sites abandoned for 4 to 6 years (table 1).

Distribution of aspen in height/DBH size-class densities was not different among the treatments, except for the sprout category (table 2). Sprout density was greatest for sites abandoned for 1 to 3 years and was different than all of the treatments except the active sites. Sprout densities did not differ for active sites, sites abandoned for 4 to 6 years, sites abandoned for 7 to 11 years, and control sites. Ungulate use of sprouts was greatest in the control sites averaging 61 percent. Ungulate browsing on saplings was more intense in control sites and sites abandoned for 7 to 11

years than in other treatments, with 69 and 54 percent respectively. The active sites and sites abandoned for 4 to 6 years had the lowest sapling use by ungulates.

DISCUSSION

Aspen stands in Eagle Creek produced large amounts of ramets after beaver disturbance, but restricted ramet growth and decreased ramet density over time suggests some limiting factor is preventing recovery. High densities of aspen sprouts and saplings were observed in active sites and in sites abandoned for 1 to 3 years (table 1). However, by the fourth year after beaver abandonment, sprout and sapling densities had decreased and were similar to non-disturbed sites. Additionally, aspen cohorts stimulated by beaver cutting did not graduate into larger size-classes. The average number of branches browsed by ungulates for each aspen sprout and sapling was $\approx$ 40 percent along Eagle Creek. This amount of use has prevented most aspen suckers from escaping the browse zone (>2m) and growing into the overstory. Beaver removed most of the existing mature stems in the disturbed sites and the suckers stimulated by this disturbance have not been able to replace them under current ungulate use.

Aspen stands can be stimulated to produce additional ramets via the removal of existing stems by beaver. Increased asexual reproduction can provide the necessary stem recruitment to ensure stand longevity. However, this process obviously requires the presence of beaver, habitat to sustain their activities, and sufficient aspen stand vigor to produce new sprouts. Given these caveats, the role of beaver in aspen stand restoration on the NYWR is currently limited and the future is uncertain. The efforts at re-establishment we monitored actually accelerated the decline in mature aspen cover.

Table 1—Comparison¹ of diameter class densities² in control sites, active sites, and sites abandoned for 1-3, 4-6, and 7-11 years in Eagle Creek, Montana.

Basal Diameter Class (stems/m ²)	Control (n = 5)	Active (n = 6)	Abandoned 1-3 Years (n = 7)	Abandoned 4-6 Years (n = 4)	Abandoned 7-11 Years (n = 5)
Sprout	0.32 ^a	1.35 ^b	1.29 ^b	0.26 ^a	0.23 ^a
Sapling	0.33 ^a	0.93 ^b	0.9 ^b	0.46 ^a	0.34 ^a
Pole	0.13	0.36	0.4	0.36	0.37
Mature	0.17 ^a	0.04 ^a	0.07 ^a	0.01 ^b	0.07 ^b
Total Stem Density	0.95 ^a	2.7 ^b	2.6 ^b	1.09 ^a	1.01 ^a

¹Different superscripts (a and b) within each row represent significant differences ($P < 0.05$).

²Densities calculated from 30-1m² plots per site.

Table 2—Comparison¹ of aspen height/DBH size-class densities² in control sites, active sites, and sites abandoned for 1-3, 4-6, and 7-11 years in Eagle Creek, Montana.

Height/DBH Size-class (stems/m ²)	Control (n = 5)	Active (n = 6)	Inactive 1-3 Years (n = 7)	Inactive 4-6 Years (n = 4)	Inactive 7-11 Years (n = 5)
Sprout	0.31 ^a	1.31 ^{ab}	1.75 ^b	0.29 ^a	0.14 ^a
Sapling	0.17	0.45	0.72	0.15	0.15
Recruitment Pole	0.14	0.18	0.04	0.14	0.15
Non-recruitment Pole	0.01	0.01	0	0	0
Mature	0.09	0.01	0	0	0.05

¹Different superscripts (a and b) within each row represent significant differences ($P < 0.05$).²Densities calculated from 60m² belt transect per site.

REFERENCES

- Bartos, D.L., Mueggler, W.F., Campbell, R.B. 1991. Regeneration of aspen on burned sites in western Wyoming. USDA, Forest Service INT-448. Intermountain Forest and Range Experiment Station, Ogden, UT.
- D'Agostino, R.B., Bangler, A., D'Agostino Jr., R.B. 1990. A suggestion for using powerful and informative tests of normality. *The American Statistician*. 44:316-321.
- DeByle, N.V. 1985. Wildlife. *Aspen: Ecology and Management in the Western United States*, edited by N.V. DeByle and R.P. Winokur. US Forest Service, General Technical Report RM-119. 135-152.
- Devore, J.L., Peck, R. 2001. *Statistics: The exploration and analysis of data*. Duxbury, Pacific Grove, CA.
- Donkor, N.T., Fryxell, J.M. 1999. Impact of beaver foraging on structure of lowland boreal forests of Algonquin Provincial Park, Ontario. *Forest Ecology and Management*. 118:83-92.
- Gallatin National Forest. 1987. Forest Plan. USDA Forest Service Bozeman, MT.
- Hann, W.J., Jensen, M.E. 1987. *Ecosystem Classification Hand book*. USDA FS Ecodata Methods. FSH 12/87 R-1 SUPP 1.
- Houston, D.B. 1982. *The Northern Yellowstone Elk, Ecology and Management*. Macmillan, N.Y.
- Johnson, C. A., Naiman, R.J. 1990. Browse selection by beaver: effects on riparian forest composition. *Journal of Forest Research*. 20:1036-1043.
- Jonas, R.J. 1955. *A Population and Ecological Study of the Beaver (Castor canadensis) of Yellowstone National Park*. M.S. Thesis, University of Idaho.
- Jones, J.R., DeByle, N.V. 1985. Morphology. *Aspen: Ecology and Management in the Western United States*, edited by N.V. DeByle and R.P. Winokur. US Forest Service, General Technical Report RM-119. 11-18.
- Kay, C.E. 1990. *Yellowstone's Northern Elk Herd: A critical Evaluation of the "Natural-Regulation" Paradigm*. Ph.D. Dissertation. Utah State University, Logan.
- Keigley, R.B., Frisina, M.R., Fager, C.W. 2002. Assessing browse trend at the landscape level Part One: Preliminary steps and field survey. *Rangeland*. 24:28-33.
- National Research Council. 2002. *Conclusions and recommendations. Ecological dynamics of Yellowstone's northern range*. National Academy of Press, Washington, D.C.. 120-137.
- Rosgen, D., Silvey, H.L. 1996. *Applied River Morphology*. Wildland Hydrology. Pagosa Springs, CO.
- Smith, D.W. 2005. Unpublished. Aerial Beaver Survey of Yellowstone National Park. National Park Service Wildlife Biologist.
- St. John, R.A. 1995. *Aspen Stand Recruitment and Ungulate Impacts*. M.S. Thesis, University of Montana. Missoula, MT.
- Stegman, L.C. 1954. Production and utilization of aspen by beaver. *Journal of Wildlife Management*. 18: 348-358.
- Warren, E.R. 1926. *A Study of Beaver in the Yancy Region of Yellowstone National Park*. Roosevelt Wildlife Annual. 1:1-191.

Sagebrush-Obligate Passerine Response to Ecological Site Characteristics

Mary I. Williams¹, Thomas L. Thurow², Ginger B. Paige³,
Ann L. Hild⁴, and Kenneth G. Gerow⁵

ABSTRACT

*Adoption of ecological sites as monitoring and management units by a variety of land users has prompted discussion of their benefits for wildlife habitat management. Density and occurrence of shrub-steppe passerines are often related to key habitat characteristics such as plant species composition, cover, and structure. Until recently, ecological sites have not been tested as units for monitoring and management of passerines. We conducted a study implementing ecological sites as management units and used passerines as indicators of potential use of these sites. Ecological site characteristics and three sagebrush-obligate passerines were quantified on ecological sites at and near Browns Park National Wildlife Refuge in Colorado. In 2006 and 2007, we surveyed passerines and site characteristics using standard techniques within 101, 100-m radius plots. Density of Brewer's sparrow (*Spizella breweri*) and occurrence of Brewer's sparrow, sage sparrow (*Amphispiza belli*), and sage thrasher (*Oreoscoptes montanus*) were estimated for six ecological sites and then related to site characteristics. For example, Brewer's sparrow densities were greatest (3.0 birds/ha) on a Loamy Fine Sand Ecological Site containing taller vegetation than vegetation for other ecological sites. Scientific literature commonly associates Brewer's sparrows with sagebrush (*Artemisia tridentata*) presence, but on ecological sites at Browns Park Brewer's densities are related more to vegetative structure rather than species composition. Results show there are links between passerine populations and ecological sites; a relationship which provides a meaningful foundation in developing long-term monitoring protocols and enhancing management decisions to favor sagebrush-obligate passerines.*

INTRODUCTION

An ecological site is characterized by the ability to produce certain types and amounts of vegetation based on climate, topography and soil, and can be practical spatial units for monitoring and management (Pellant and others 2005; Herrick and others 2006a). Ecological site descriptions

serve as repositories of geographic locations, climate, topography, soils, hydrology, vegetation communities, and state and transition models. State and transition models offer a tool for defining what is possible and realistic for management objectives, options and priorities (Bestelmeyer and others 2003; Herrick and others 2006a). Ecological sites are practical units because they are consistent with current monitoring, management and assessment frameworks used by land management agencies.

Adoption of ecological sites as basic management units by the U.S.D.A. Natural Resource Conservation Service (NRCS) and U.S. Bureau of Land Management (BLM) has prompted discussion in their application for wildlife habitat management and research. Units based on ecological site delineations enable land managers to evaluate not only potential and health of each site within a landscape, but also to examine management options and successes. Incorporation of qualitative and quantitative data for sites in a landscape will help determine where and how management will have the greatest impact on resources such as wildlife habitat. Recent suggestions are to expand ecological site descriptions to include vegetation and site characteristics that reflect wildlife habitat parameters such as plant distribution and structure. Ecological site descriptions by definition provide information on potential plant communities for a given soil in a particular climate; these soil and plant community characteristics dictate wildlife habitat.

Shrub-steppe passerine populations, particularly sagebrush- (*Artemisia tridentata*) obligate species, are an appropriate starting point for evaluating ecological sites as monitoring and management units for wildlife habitat because they are frequently monitored, often discussed within the context of other species of concern (for example, sage-grouse (*Centrocercus* spp.)), sensitive to anthropogenic and natural events, and have a tight association with shrublands. Sagebrush-obligate passerines, namely Brewer's sparrow (*Spizella breweri*), sage sparrow (*Amphispiza belli*), and sage thrasher (*Oreoscoptes montanus*) require sagebrush for some part of their lifecycle, mostly for nesting and shelter (McAdoo and others 2004). They have been studied extensively over the past 40 years. Despite the wealth of data, few studies have addressed sagebrush-obligate population parameters (abundance, density, occurrence, occupancy, etc.) in relation to ecological sites. This is partly due to challenges regarding scale and lack of information on linkages between ecological site characteristics and population parameters.

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹PhD Candidate, Rangeland Ecology and Watershed Management, Department of Renewable Resources, University of Wyoming, Laramie. mwillia9@uwyo.edu

²Professor, Range and Forest Watershed Management, Department of Renewable Resources. tthurow@uwyo.edu

³Assistant Professor, Water Resources, Department of Renewable Resources University of Wyoming, Laramie. gpaige@uwyo.edu

⁴Professor, Rangeland and Shrubland Ecology, Department of Renewable Resources. annhild@uwyo.edu

⁵Professor, Biometry, Department of Statistics, Laramie. gerow@uwyo.edu

Passerine populations fluctuate across time and space leading to mixed results in developing habitat association models (Rotenberry and Wiens 1980; Rotenberry and Knick 1999). Scales at which species parameters are defined have an effect on how habitat associations are interpreted (Wiens and Rotenberry 1981; Wiens and others 1987; Knick and Rotenberry 2000). Sage thrashers were positively associated with shrub and bare ground cover, vegetation height and habitat heterogeneity and negatively related to grass and low vegetation cover at broad spatial scales (tallgrass prairie to shrubsteppe regions of U.S.), but were inversely related to sagebrush and total shrub cover at regional scales (shrub-steppe of the Pacific Northwest) and positively related to standing dead vegetation and negatively related to shrub cover at the individual scale (territories within 9 ha plots) (Wiens and others 1987). Passerine parameters are often assessed in areas that are much larger than their breeding territories. Brewer's sparrow territories range from 0.1 to 2.4 ha (Paige and Ritter 1999; Rotenberry and others 1999; and Wiens and others 1985). Sage sparrow breeding territories range from 0.6 to 6.3 ha in shrub-steppe (Paige and Ritter 1999; Reynolds 1981; Rich 1980; Wiens and others 1985) and sage thrasher ranges from 1.1 to 1.9 ha (Reynolds 1981). Ecological sites are groupings of similar soils based on their potential to support certain types and amounts of vegetation and are mapped using second to third order soil surveys, which range between 1:12 000 to 1:63 000 in scale (for example, Soil Survey Geographic Database, SSURGO). Sites occur together in landscapes and generally repeat across landscapes. Therefore, an ecological site may include several potential passerine breeding territories and an individual territory may expand over more than one ecological site. Knowing the scales of both passerine breeding and habitat territories is essential for integration with ecological sites across a landscape.

The lack of information on linkages between site characteristics and passerine population parameters could be reflective of monitoring protocols and an insufficient number of studies conducted on ecological sites. Brewer's sparrow, sage sparrow and sage thrasher prefer similar breeding habitats – above average vegetation height, high shrub and bare ground cover, and low grass cover (Paige and Ritter 1999). Shrub species composition is also an important habitat characteristic for these species (Wiens and Rotenberry 1981). In addition, reductions in live shrub cover (Best 1972; Reynolds and Trost 1981; Wiens and Rotenberry 1985) and patch size (Knick and Rotenberry 1995; Vander Haegen and others 2000) have a negative effect on population size. Plant species composition, cover and structure measurements are common to passerine studies and are occasionally collected using standard monitoring methods which make it difficult for comparison and application across spatial scales. Some studies have shown that wildlife populations vary with soil types (Dobler and others 1996; Vander Haegen and others 2000). Vander

Haegen and others (2000) found strong relationships between sagebrush-obligate passerine parameters and soil type; populations were associated with deep, loamy soil communities that supported greater cover of tall shrubs than shallow soils. Passerine association with soil type is a strong indicator that ecological sites (soil-vegetation associations) would be a valuable structure for assessing habitat characteristics and resource use for these populations.

Using an integrated framework we delineated and described upland ecological sites on a wildlife refuge in northwestern Colorado and linked site characteristics with sagebrush-obligate passerine density and occurrence. We used common, standard monitoring methods to measure passerines and site characteristics within the context of ecological sites. Our objectives were to 1) delineate and describe upland ecological sites, 2) survey passerines and intensively measure site characteristics within each site, 3) relate density and occurrence to site characteristics, and 4) determine how ecological site characteristics relate to observed density and occurrence of passerines. Studying habitat characteristics within an ecological site context can help identify site characteristics as potential habitat indicators for passerines. Illustrating that ecological sites are reflective of potential use by passerines provides the opportunity for researchers, land agencies and land owners to not only communicate in the same language, but also to determine management practices and provide a framework to evaluate management success.

MATERIALS AND METHODS

Study Area

The study was conducted on Browns Park National Wildlife Refuge (BPNWR) and on adjacent areas owned by BLM and State of Colorado in northwestern Colorado near Maybell. The area, approximately 5445 ha with an average elevation of 1,633 m, lies within the Upper Colorado River Ecosystem. Average annual temperature is 7.4 °C, with average annual winter, spring and summer temperatures of -3.7, 7.2 and 18.6 °C, respectively. Annual precipitation is 216 mm, half of which occurs in winter and spring as frontal storms. Soils are formed by alluvial and eolian processes from hard sandstone, limestone, and small amounts of quartzite. They range from loamy fine sands to sandy loams supporting a mosaic of upland shrub-steppe vegetation that covers more than 60 percent of the study area. Shrubs and perennial grasses common to both sagebrush-steppe and salt-desert plant communities are present and include Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*, hereafter referred as sagebrush), shadscale (*Atriplex confertifolia*), spiny hopsage (*Grayia spinosa*), black greasewood (*Sarcobatus vermiculatus*), Indian ricegrass (*Achnatherum hymenoides*)

and needleandthread (*Hesperostipa comata*). Primary land uses are wildlife habitat, livestock grazing and recreation. The refuge was established in 1963 by U.S. Fish and Wildlife Service (FWS) to provide sanctuary for migratory birds, conserve threatened and endangered species, and enhance species' habitat. Winter livestock grazing ceased on the refuge in 1994 but continues on adjacent BLM and State lands.

Ecological Sites

Spatial data of the study area, which included Browns Park National Wildlife Refuge and surrounding areas, were integrated into a geographic information system (GIS) to identify upland ecological sites. Soil information was obtained from SSURGO for Moffat County, Colorado (Soil Survey Staff 2006). Supporting spatial data (vegetation, water features, roads, and elevation) were obtained from Browns Park National Wildlife Refuge. The area is within the 7 to 9 inch precipitation zone of NRCS Major Land Resource Area 34A-Central Desertic Basins, Mountains and Plateaus which extends into Wyoming and Utah. Because ecological sites had not been delineated or described we collected additional baseline information including onsite soil evaluations and data from similar ecological sites in 34A. We delineated six ecological sites based on soil profile and surface textures. Loamy Fine Sand sites occur in both upland and bottomland areas and have loamy fine sand surface and subsurface soil textures; the soils are slightly alkaline. The remaining five sites are upland sites. Upland Cobbly Sand sites have a loamy sand surface with cobble sand subsurface. A cobble layer exists approximately 4 cm deep in the profile. Upland Gravelly Sand sites have a loamy fine sand surface with gravelly loamy fine sand subsurface textures. Upland Loamy Fine Sand sites have a loamy fine sand surface and subsurface textures with a possible calcium carbonate subsurface layer indicated by presence of winterfat (*Krascheninnikovia lanata*) (Blaisdell and Holmgren 1984). Upland Sands sites have a fine sand surface texture and loamy fine sand with occasional cobbles in subsurface layers. Site characteristics and passerines were measured in 101 plots randomly stratified within each ecological site at 250 m spacing (figure 1).

Site Characteristics

Standard monitoring methods adapted from Herrick and others (2005a, 2005b) were selected to measure site characteristics that would capture ecological site variability and indicate potential passerine density and occurrence. Site characteristics were measured in all plots along three 50 m line transects arranged in a spoke design between May and August 2006 and 2007.

Type and amount of cover (percent) and vegetation height (cm) were measured by line point intercept methods. Shrub density (shrubs/ha) was measured by means of a 2 by 50 m

belt transect. Shrubs were grouped into four size categories (A < 10, B 10 to 50, C 50 to 100, and D > 100 cm in height). Amount and size of gaps (> 20 cm) between vegetation canopies and bases were measured by gap intercept methods. A qualifying canopy was defined as at least 50 percent of the plant canopy covering no less than 3 cm along the line. A qualifying base had to be large enough to interrupt a linear flow pattern. To further quantify shrub structure, the amount, size and area of intercanopy spaces between shrub bases and canopies, were also measured using gap intercept methods with height measurements. Canopy and basal gaps were calculated separately as the percentage of line covered in gaps 20 to 50, 51 to 100, 101 to 200 and > 200 cm in length. All measurements were averaged across the three transects for each plot. GPS locations and photographs were taken of each plot.

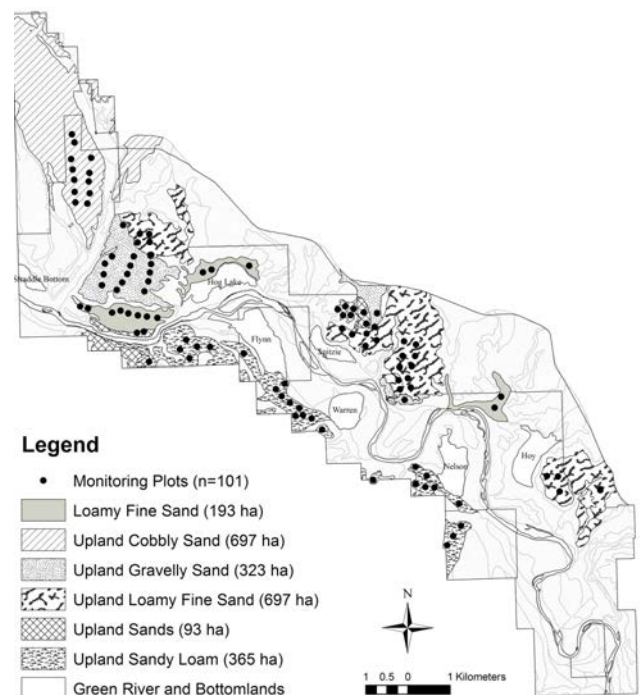


Figure 1—Upland ecological sites (number of ha) and monitoring plots at the Browns Park study area located 60 miles northwest of Maybell, Colorado. Sites were delineated by soil texture characteristics.

Passerine Density and Occurrence

Upland passerines were surveyed in each plot during the breeding season using fixed-radius point transect sampling methods (Bibby and others 2000 and Buckland and others 2001). Survey dates were May 6 to 11 and May 26 to 31, 2006 and May 6 to 11 and 23 to 27, 2007. One observer (same individual) surveyed passerines on mornings between 0600 and 1000 hours with no precipitation and wind less than 20 km/h. Male passerines were identified by sight or

sound within 100 m of each plot for 5 minutes following a 1 to 2 minute settling period. For each detected male passerine, species was recorded and distance estimates were made with a laser range finder. Although all passerine species were recorded, we are limiting analyses to Brewer's sparrow, sage sparrow and sage thrasher.

Densities were estimated with program DISTANCE, version 5.0, release 2 (Thomas and others 2005). DISTANCE uses perpendicular distances of each observation to create a histogram of the number of detections by distance. A series of key functions (half-normal, hazard and uniform) with possible hermite and simple polynomial and cosine adjustments were used to model detection functions for each species. Prior to modeling, data were pooled across survey dates and at least 10 percent of the largest distances were truncated as

recommended by Buckland and others (2001). Model fit was evaluated with a chi-square goodness-of-fit (GOF) test and Akaike's Information Criterion with a second order correction for small sample size (AICc) was used to select the most robust model (Buckland and others 2001). Through modeling a detection function, an estimate of the proportion of individuals detected in the survey area was derived using standard distance estimation methods for point transects (Buckland and others 2001; Buckland and others 2010).

Occurrence (percent) by ecological site was estimated for each species and defined as the number of plots where the species was present divided by the number of plots on each site. A species was considered present in a plot if it occurred in at least one survey in any year.

Table 1—Site characteristic means (standard error) for six upland ecological sites at the Browns Park study area. Within each site characteristic, ecological site means with followed by the same letter are not statistically different (Fisher's protected least significant difference, $p \geq 0.05$).

Site Characteristic	Loamy Fine Sand (11) ^y	Upland Cobbly Sand (12)	Upland Gravelly Sand (19)	Upland Loamy Fine Sand (24)	Upland Sands (10)	Upland Sandy Loam (25)
Cover (%)						
Sandberg bluegrass	0.1 (0.1)b	1.1 (0.5)a	0 (0)b	0.1 (0.1)b	0 (0)b	0.1 (0.03)b
Greasewood	23.9 (3.8)a	12.1 (1.9)b	0.2 (0.1)d	3.9 (0.7)cd	1.1 (0.5)d	7.9 (2.3)bc
Rabbitbrush	0.5 (0.4)b	0.2 (0.2)b	1.3 (0.5)b	0.3 (0.2)b	3.4 (2.1)a	0.1 (0.1)b
Shadscale	1.9 (0.8)c	2.7 (0.5)c	9.7 (1.0)a	5.5 (0.7)b	2.3 (0.6)c	1.1 (0.3)c
Spiny hopsage	2.4 (1.2)bc	0.1 (0.1)c	0.3 (0.3)c	2.7 (0.9)b	7.3 (1.9)a	3.6 (0.6)b
Winterfat	0 (0)b	0 (0)b	0 (0)b	2.3 (1.0)a	0.1 (0.1)b	0 (0)b
WY big sagebrush	1.8 (1.4)bc	8.8 (2.1)a	8.5 (2.2)a	0.9 (0.7)c	5.7 (1.2)ab	4.1 (0.9)bc
Total shrub	30.9 (3.1)a	23.9 (1.7)ab	20.1 (2.3)bc	15.7 (1.1)c	20.4 (2.0)bc	17.2 (2.0)c
Interspace litter	17.8 (1.1)b	16.8 (1.5)b	20.6 (0.9)ab	20.7 (1.3)ab	18.6 (1.8)b	23.6 (1.5)a
Height (cm)^z						
Vegetation	26.3 (1.9)a	21.7 (1.1)b	15.2 (0.9)c	15.2 (0.9)c	18.7 (1.2)bc	20.9 (1.5)b
Shrub	30.5 (2.1)a	24.0 (1.0)bc	20.2 (1.4)c	20.8 (0.9)c	25.4 (2.8)ab	28.3 (1.4)a
Shrub Gap	45.6 (2.5)a	32.0 (1.0)cd	26.2 (0.9)d	26.8 (1.3)d	36.0 (2.2)bc	39.2 (2.5)b
Canopy Gaps (%)						
All 20-50 cm	4.4 (0.6)c	6.0 (0.4)abc	7.8 (0.6)a	7.3 (0.8)ab	7.8 (0.8)a	5.8 (0.8)bc
All 51-100 cm	9.6 (1.0)c	12.4 (0.6)bc	14.6 (0.9)ab	15.2 (1.1)ab	16.5 (1.2)a	11.8 (1.2)c
Shrub 20-50 cm	3.7 (0.5)bc	5.0 (0.4)ab	5.7 (0.7)a	4.3 (0.7)ab	3.4 (0.6)bc	2.5 (0.3)c
Shrub 51-100 cm	8.3 (0.8)bc	10.7 (0.6)ab	12.2 (1.4)a	11.3 (1.2)ab	8.7 (0.9)bc	7.0 (0.8)c
Shrub 101-200 cm	16.9 (1.6)ab	21.4 (1.4)a	19.4 (1.7)a	19.2 (1.1)a	16.7 (1.3)ab	14.6 (1.1)bc
Shrub > 200 cm	28.4 (4.7)bc	26.0 (2.7)bc	24.8 (3.6)c	35.6 (2.9)b	37.4 (3.6)ab	45.0 (2.8)a
Shrub Density (no/ha)						
A (< 10 cm)	851.6 (320.0)b	3255.5 (635.7)a	3420.6 (568.0)a	2866.4 (598.4)a	1250.0 (289.8)b	1268.0 (369.1)b
B (10-50 cm)	5966.6 (1535.7)c	15394.4 (1430.7)ab	18629.1 (2011.4)a	17333.3 (2666.6)a	9880.0 (1620.4)bc	8019.9 (1574.4)c
C (50-100 cm)	4724.3 (398.7)a	2983.3 (316.3)b	1022.1 (176.0)c	1456.9 (234.3)c	2700.0 (357.0)b	3345.3 (398.6)b
D (> 100 cm)	327.2 (109.1)a	16.6 (8.7)bc	7.0 (4.1)c	9.7 (5.1)c	80.0 (35.2)bc	131.9 (54.2)b

^yEcological site (number of plots).

^zVegetation and shrub height measured with line point intercept methods. Shrub gap height measured with gap intercept methods.

Data Analysis

Site characteristics were assessed for differences among ecological sites using univariate analysis of variance (ANOVA). Residuals were assessed for meeting the assumptions of ANOVA. As a result of heteroscedasticity, sagebrush, greasewood, rabbitbrush (*Chrysothamnus viscidiflorus*), shadscale, spiny hopsage, and winterfat cover and density of small (A and B) and large (D) shrubs were natural log transformed. Results from analyses of original data did not change with transformation. Where significant differences occurred among sites ($\alpha = 0.05$), mean separations were conducted using Fisher's protected least significant difference (FLSD).

Differences in ecological site passerine density were assessed using ANOVA. Multiple linear regression was used to examine relationships between density and site characteristics and differences in density among ecological sites. Prior to performing regressions, Pearson correlation coefficients were generated to select site characteristics that were significantly related to species density because the original data set included over 60 site characteristics. Best subsets regressions were conducted using sagebrush and total shrub cover, vegetation height, shrub density, and species-specific site characteristics to select the best one, two and three predictor models for density. Adjusted R^2 , Mallows Cp, and parameter standard errors were evaluated to select the best set of models with up to three site characteristics. Models were further evaluated with respect to multicollinearity and sampling logistics. Comparisons of occurrence across ecological site were made for each species, as well as evaluation of occurrence for species on each ecological site.

RESULTS

Ecological Site Characteristics

As expected, ecological sites differed in several site characteristics (table 1). Major differences were found in shrub cover and density among sites, while there were also some significant differences in height and canopy gaps. Loamy Fine Sand had greater density and cover of large shrubs, especially greasewood, whereas Upland Gravelly Sand was characterized by greater cover of low shrubs such as shadscale and shorter overall vegetation. Upland Cobbly Sand was distinguished by having several intermediate shrub canopy gaps (101 to 200 cm) and greater sandberg bluegrass (*Poa secunda*) and sagebrush cover. Upland Loamy Fine Sand had the greatest cover of winterfat relative to all other ecological sites. Upland Sands had greater spiny hopsage and rabbitbrush cover and Upland Sandy Loam had greater interspace litter and several, large shrub canopy gaps (> 200 cm).

Passerine Density and Occurrence

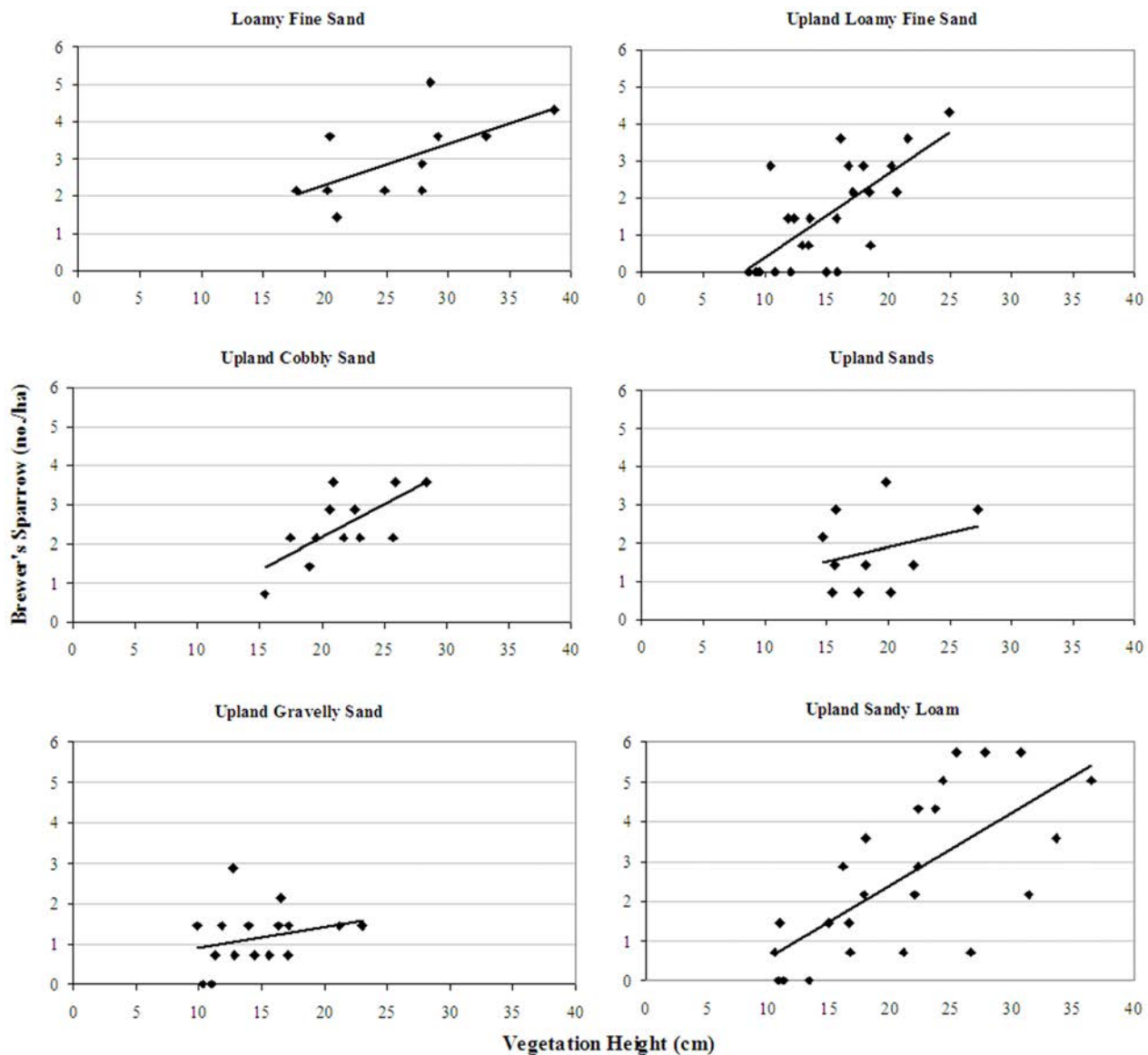
Including three sagebrush-obligates (Brewer's sparrow, sage sparrow and sage thrasher), 25 species and 989 male passerines were observed over the two years. The most common species observed was Brewer's sparrow (30 percent of all observations and 2.02 birds/ha over the entire study area). Sage sparrows and sage thrashers were detected in 9 and 4 percent of all observations (0.19 and 0.09 birds/ha over the entire study area, respectively). We recorded densities of Brewer's sparrow, sage sparrow and sage thrasher similar to other shrub-steppe communities (Boyle and Reeder 2005; Rotenberry and others 1999). Greater density of Brewer's sparrow than sage sparrow and sage thrasher is consistently seen in the shrub-steppe (Paige and Ritter 1999). Of the sagebrush-obligates, only Brewer's sparrow observations were sufficient to estimate plot density.

Brewer's sparrow densities differed among ecological sites and ranged from 1.2 to 3.0 birds/ha. Brewer's sparrows were denser and greater in Loamy Fine Sand than in Upland Sands, Upland Loamy Fine Sand and Upland Gravelly Sand sites. Brewer's sparrow densities were regressed against the following site characteristics: sagebrush and total shrub cover, vegetation height, shrub density, and vegetation and shrub canopy and basal gaps. Three regression models generated from best subsets were selected (table 2). In the first model, Brewer's sparrow density increased with increasing vegetation height and reduced presence of small gaps between plant bases and density of small shrubs (accounting for 53 percent of variation in density). For the second model, density increased with increasing vegetation height and reduced presence of small gaps between plant canopies, explaining 50.8 percent of the variance in density. Using a single site characteristic model, vegetation height, explained 48.5 percent of the variation in density. Of all site characteristics, vegetation height had the strongest independent correlation to density ($r = 0.70$). Vegetation height was negatively correlated to small canopy and basal gaps and density of small shrubs ($r > -0.40$). Densities were strongly related to vegetation height on ecological site (figure 2). Overall, Brewer's sparrows were denser in ecological sites dominated by taller vegetation. The relationship between density and vegetation height was not as strong in the Upland Gravelly Sand and Upland Sands sites.

Brewer's sparrow was present more than 70 percent of the plots across all ecological sites, but least in Upland Loamy Fine Sand (figure 3). Both sage sparrows and sage thrashers were less common overall, but their occurrences were greatest in Upland Cobbly Sand and Upland Sands sites. Sage thrashers were not recorded in Upland Gravelly Sand.

Table 2—Best subset regression models for Brewer's sparrow density (birds/ha).

Model	Coefficient	Standard Error	P	Adj R^2
Vegetation Height (cm) ^a	0.12	0.02	0.000	53.0
All Basal Gaps 51-100 cm (%)	-0.06	0.03	0.024	
B (10-50 cm) ^b	-3.61×10^{-5}	1.16×10^{-5}	0.002	
Constant	0.54	0.56	0.335	
Vegetation Height (cm)	0.14	0.02	0.000	50.8
All Canopy Gaps 51-100 cm (%)	-0.05	0.02	0.021	
Constant	0.18	0.59	0.766	
Vegetation Height (cm)	0.16	0.02	0.000	48.5
Constant	-0.99	0.33	0.003	

^aVegetation height measured by line point intercept methods.^bDensity of shrubs between 10 and 50 cm in height.**Figure 2**—Brewer's sparrow density and vegetation height for Loamy Fine Sand ($n = 11$), Upland Cobbly Sand ($n = 12$), Upland Gravelly Sand ($n = 19$), Upland Loamy Fine Sand ($n = 24$), Upland Sands ($n = 10$), and Upland Sandy Loam ($n = 25$) ecological sites. Lines are fitted regression line of density and height on each site.

DISCUSSION

Variation and similarities among and on sites at Browns Park coupled with results from passerine observations point toward the importance of ecological site proximity and size in developing long-term monitoring and management protocol for a landscape. Sites existing in small proportions or in small patches within the entire landscape may not be given high monitoring or management priority because of operation constraints. Whereas, management options may be evaluated for large, adjacent ecological sites with similar soils such as Upland Loamy Fine Sand and Upland Gravelly Sand, to improve sagebrush-obligate passerine habitat, which could potentially benefit other sagebrush-obligate species.

Differences in Brewer's sparrow densities among ecological sites showed a strong response to plant community structure rather than plant species composition. Brewer's sparrow is commonly and consistently associated with shrublands dominated by sagebrush (Paige and Ritter 1999), but in this study, greater densities were associated with sites with tall, widely spaced vegetation. The negative influence of small plant canopy and basal gaps on species density is notable as gap size is a common indicator measured on ecological sites. Gaps between plant canopies and bases are important indicators of resistance to runoff, water and wind erosion, and invasive species establishment (Pellant and others 2005). However, because of multicollinearity between vegetation height, gaps and shrub density it is difficult to interpret their individual effects on observed densities for each site. Brewer's densities were influenced strongly by vegetation height, especially in sites with mean heights greater than 20 cm. Relationships between density and height were inconsistent in sites where densities were low. Mean density and vegetation height did not differ among Upland Gravelly Sand, Upland Loamy Fine Sand and Upland Sands, but there was a stronger linear relationship between density and height on Upland Loamy Fine Sand than the two other sites. For Upland Sands, this was attributed to a smaller mapped area and number of plots relative to other sites at Browns Park. In both sites, however, Brewer's sparrow may be responding to site characteristics other than vegetation height such as small canopy and basal gaps and density of small shrubs. Monitoring of additional Upland Sands and Upland Gravelly Sand sites within MLRA 34A will help to distinguish sample size from site characteristic effects on Brewer's sparrow density. Measuring vegetation height using line point intercept methods was easier and faster than measuring gaps and shrub density. As revealed by the coefficient of determination, vegetation height could provide as much information as gap size and shrub density in understanding densities found on sites.

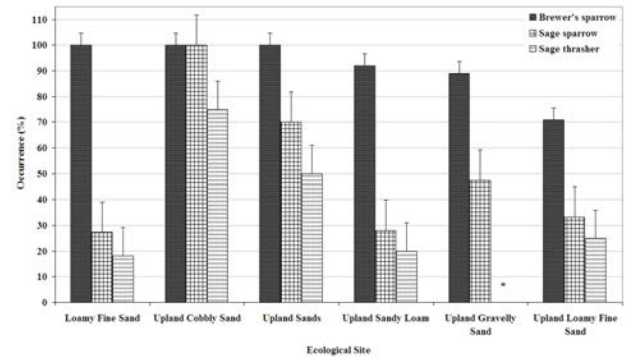


Figure 3—Occurrence (± 1 standard error) of Brewer's sparrow, sage sparrow and sage thrasher for Loamy Fine Sand ($n=11$), Upland Cobbly Sand ($n=12$), Upland Gravelly Sand ($n=19$), Upland Loamy Fine Sand ($n=24$), Upland Sands ($n=10$), and Upland Sandy Loam ($n=25$) ecological sites.

Vegetation height reflects soil characteristics and past management efforts. Recognizing that height is a meaningful indicator of potential use by Brewer's sparrow encourages inclusion of height measurements in ecological site descriptions, monitoring protocols and evaluation of sites and management options that favor this species. For instance, Upland Loamy Fine Sand includes two soil types, one having a minor component with elevated amounts of calcium carbonate in surface and subsurface layers. Three monitoring plots in this site are dominated by winterfat (9-20 percent), a low growing shrub rarely exceeding 10 cm in height at this site. The dominance of winterfat certainly contributes to structural characteristics of the vegetation – low vegetation height, high density of small shrubs and several small plant gaps – characteristics not favored by Brewer's sparrow as there were no detections for these plots. Long-term monitoring will help detect any changes to vegetation height, but will also help determine if these winterfat-dominated areas on the site are exhibiting a plant community state or a transition to another, separate ecological site.

Brewer's sparrows occurred in all sites, indicating a ubiquitous pattern across the study area. However, observed densities varied among ecological sites indicating and point toward existence of primary and secondary habitats on sites. All three sagebrush-obligate species frequented sites with high cover of tall shrubs (Upland Cobbly Sand and Upland Sands), in contrast, sage sparrow and sage thrasher had low occurrence in sites with greater cover of greasewood and spiny hopsage and lower cover of sagebrush. This pattern implies that sage sparrow and sage thrasher prefer sites dominated or co-dominated by sagebrush, which is consistent with literature. Sage sparrows have similar breeding habitat preferences as Brewer's sparrow, but are perhaps more obligate to sagebrush, preferring large, continuous stands of sagebrush (Rich 1978). Affinity for sagebrush is less straightforward for the sage thrasher,

especially on Upland Gravelly Sand where sagebrush cover is higher than other sites but sage thrashers were never observed. Presence of sagebrush cover alone therefore does not indicate potential use by sage thrasher. Sage thrashers are almost always associated with shrublands dominated by sagebrush and prefer areas with more shrub and bare ground cover and less grass and spiny hopsage cover (Rotenberry and Wiens 1980; Wiens and Rotenberry 1981). In this study, occurrence of sage thrasher in Upland Cobbly Sand and Upland Sands indicates a preference for sagebrush height rather than its presence. Occurrence patterns indicate a difference among the three sagebrush-obligates in response to ecological site characteristics and illustrate that it is important to not only measure plant composition but also community structure. Upland Gravelly Sand may never provide suitable habitat for sage thrashers even with management activities, whereas Upland Sandy Loam has potential for improving the existing plant community to favor sage sparrow and sage thrasher.

Using sagebrush-obligate passerines as indicators of potential use of ecological sites present in northwestern Colorado provided an opportunity to study these populations in a diverse shrub community where sagebrush is not the dominant shrub species. The patterns of sagebrush-obligate density and occurrence indicate that shrub community composition is driving their distribution more than presence of a shrub species. Baseline data coupled with continued monitoring and state and transition models for each site will aid determining management options.

CONCLUSIONS

Response from sagebrush-obligate passerines to ecological sites and site characteristics at Browns Park suggests that ecological sites are a practical framework as wildlife management units. For the refuge, having these ecological sites mapped and described within a spatial context will be useful for making landscape scale management decisions that favor sagebrush-obligate passerines. Monitoring and management should be prioritized according to ecological site potential, representative size and proximity to ecological sites favored by sagebrush-obligate passerines. Some sites have a wide range of variability in site characteristics that are potentially attractive to sagebrush-obligates, whereas other sites have a narrow range of variability and thus less attractive. Long-term monitoring of site characteristics using appropriate methods will aid in determining site range of variability, assessing plant community states, and evaluating management success. Further, monitoring other wildlife species on ecological sites and incorporation of sites within a landscape will significantly improve functionality of this framework.

ACKNOWLEDGEMENTS

We thank S. Beauchaine, R. Laubhan, and W. King for initiating this project and continued support. Great appreciation goes to F. Bartlett, K. Bergum, B. Chester, M. Church, J. Edwards, K. Linder, and M. Wolvington for field data collection and J. Beck, S. Early, C. Leingang, S. Miller, N. Shaw, and N. Wolf for advice and direction. Special thanks to staff at Browns Park National Wildlife Refuge, BLM (Craig Field Office), Colorado State Land Board (Moffat County), Vermillion Livestock Company, and NRCS (Colorado, Utah and Wyoming Field Offices). Funding was provided by USGS USFWS, the Department of Renewable Resources and the Graduate School at the University of Wyoming.

REFERENCES

- Bibby, C.J., Burgess, N.D., Hill, D.A., Mustoe, S.H. 2000. Bird census techniques. London: Academic Press. 302 p.
- Best, L.B. 1972. First-year effects of sagebrush control on two sparrows. *Journal of Wildlife Management* 36:534-544.
- Bestelmeyer, B.T., Brown, J.R., Havstad, K.M., Alexander, R., Chavez, G., Herrick, J.E. 2003. Development and use of state and transition models for rangelands. *Journal of Range Management* 56:114-126.
- Blaisdell, J.P., Holmgren, R.C. 1984. Managing Intermountain rangelands: salt-desert shrub ranges. General Technical Report INT-163. Ogden, UT: USDA, Forest Service, Intermountain Forest and Range Experiment Station. 52 p.
- Boyle, S.A., Reeder, D.R. 2005. Colorado sagebrush: a conservation assessment and strategy. Grand Junction, CO: Colorado Division of Wildlife.
- Buckland, S.T., Anderson, D.R., Burnham, K.P., Laake, J.L., Borchers, D.L., Thomas, L. 2001. Introduction to distance sampling: estimating abundance of biological populations. New York: Oxford University Press, Inc. 432 p.
- Buckland, S.T., Russell, R.E., Saab, V.A., Block, W.M. 2010. Analysing designed experiments in distance sampling. *Journal of Agriculture, Biology, and Environmental Statistics*. 14:423-444.
- Dobler, F.C., Eby, J., Perry, C., Richardson, S., Vander Haegen, M. 1996. Status of Washington's shrub-steppe ecosystem: extent, ownership, and wildlife/vegetation relationships. Shrub-steppe research project. Olympia, WA: Washington Department of Fish and Wildlife. 47 p.
- Herrick, J.E., Bestelmeyer, B.T., Archer, S., Tugel, A.J., Brown, J.R. 2006a. An integrated framework for science-based arid land management. *Journal of Arid Environments* 65:319-335.
- Herrick, J.E., Schuman, G.E., Rango, A. 2006b. Monitoring ecological processes for restoration projects. *Journal for Nature Conservation* 14:161-171.
- Herrick, J.E., Van Zee, J.W., Havstad, K.M., Burkett, L.M., Whitford, W.G. 2005a. Monitoring manual for grassland, shrubland and savanna ecosystems. Volume I: Quickstart. Las Cruces, NM: USDA-ARS Jornada Experimental Range. 36 p.
- Herrick, J.E., Van Zee, J.W., Havstad, K.M., Burkett, L.M., Whitford, W.G. 2005b. Monitoring manual for grassland, shrubland and savanna ecosystems. Volume II: Design, supplementary methods and interpretation. Las Cruces, NM: USDA-ARS Jornada Experimental Range. 200 p.
- Knick, S.T., Rotenberry J.T. 1995. Landscape characteristics of fragmented shrubsteppe habitats and breeding passerine birds. *Conservation Biology* 9:1059-1071.
- Knick, S.T., Rotenberry J.T. 2000. Ghosts of habitats past: contribution of landscape change to current habitats used by shrubland birds. *Ecology* 81:220-227.
- McAdoo, J.K., Longland, W.S., Evans, R.A. 2004. Nongame bird community responses to sagebrush invasion of crested wheatgrass seedings. *Journal of Wildlife Management* 53(2):494-502.

- Paige, C., Ritter, S.A. 1999. Birds in a sagebrush sea: managing sagebrush habitats for bird communities. Partners in Flight Western Working Group. Boise, ID.
- Pellant, M., Shaver, P., Pyke, D.A., Herrick, J.E. 2005. Interpreting indicators of rangeland health, version 4. Technical Reference 1734-6. USDI, Bureau of Land Management, National Science and Technology Center, Denver, CO. BLM/WO/ST-00/001+1734/Rev05. 122 pp.
- Reynolds, T.D. 1981. Nesting of sage thrasher, sage sparrow, and Brewer's sparrow in southeastern Idaho. Condor 83:61-64.
- Reynolds, T.D., Trost, C.H. 1981. Grazing, crested wheatgrass and bird populations in southeastern Idaho. Northwest Science 55:225-234.
- Rich, T.D. 1978. Cowbird parasitism of sage and Brewer's sparrows. Condor 80:348.
- Rich, T.D. 1980. Nest placement of sage thrashers, sage sparrows and Brewer's sparrows. Wilson Bulletin 92:362-368.
- Rotenberry, J.T., Knick, S.T. 1999. Multi-scale habitat associations of the sage sparrow: implications for conservation biology. Studies in Avian Biology 19:95-103.
- Rotenberry, J.T., Patten, M.A., Preston, K.L. 1999. Brewer's sparrow (*Spizella breweri*). In The Birds of North America, No. 390, edited by A. Poole and F. Gill. Philadelphia: The Birds of North America, Inc.
- Rotenberry, J.T., Wiens, J.A. 1980. Habitat structure, patchiness, and avian communities in North American steppe vegetation: a multivariate analysis. Ecology 61:1228-1250.
- Soil Survey Staff, Natural Resources Conservation Service, USDA. Soil Survey Geographic (SSURGO) Database for Moffat County, CO. Available URL: "<http://soildatamart.nrcs.usda.gov>" [Accessed 15/01/2006].
- Thomas, L., Laake, J.L., Derry, J.F., Buckland, S.T., Borchers, D.L., Anderson, D.R., Burnham, K.P., Strindberg, S., Hedley, S.L., Burt, M.L., Marques, F.F.C., Pollard, J.H., Fewster, R.M. 2005. Distance 5.0 release 2 software. University of St. Andrews, UK: Research Unit for Wildlife Population Assessment.
- Vander Haegen, W., Dobler, F.C., Pierce, D.J. 2000. Shrubsteppe bird response to habitat and landscape variables in Eastern Washington, USA. Conservation Biology 14(4):1145-1160.
- Wiens, J.A., Rotenberry, J.T. 1981. Habitat associations and community structure of birds in shrubsteppe environments. Ecological Monographs 51:21-41.
- Wiens, J.A., Rotenberry, J.T., Van Horne, B. 1985. Territory size variations in shrubsteppe birds. Auk 102:500-505.
- Wiens, J.A., Rotenberry, J.T., Van Horne, B. 1987. Habitat occupancy patterns of North American shrubsteppe birds: the effects of spatial scale. Oikos 48:132-147.

Elk and Fire Impacts on Mountain Big Sagebrush Range in Yellowstone

Carl L. Wambolt¹ and Reyer J. Rens²

ABSTRACT

Range recovery after fire has many implications for wildlife habitat. It was our objective to determine the effect of elk herbivory on recovery of mountain big sagebrush (*Artemisia tridentata* vaseyana) – bluebunch wheatgrass (*Agropyron spicatum*) – Idaho fescue (*Festuca idahoensis*) winter range a decade after a 1988 wildfire on the Black-tailed Deer Plateau of the Northern Yellowstone Winter Range. Our hypotheses were that shrubs and herbaceous plants will recover from fire equally with or without elk herbivory. Measurements were taken in and out of exclosures established in 1957 and 1962 on environmentally paired, protected and browsed-grazed sites ($n = 12$). Mountain big sagebrush cover ($n = 5$) averaged 20 percent with protection and 9.7 percent where browsed ($P \leq 0.01$). Mountain big sagebrush densities were not different ($P \leq 0.01$). The sprouting shrubs, rubber rabbitbrush (*Chrysothamnus nauseosus*), green rabbitbrush (*C. viscidiflorus*), and gray horsebrush (*Tetradymia canescens*), ($n = 4$) responded the same as mountain big sagebrush with cover of 5.8 percent and two percent where protected and browsed, respectively ($P \leq 0.001$), with no overall difference in density ($P \leq 0.47$). Few cover differences existed between individual pairs of protected and unprotected sites for herbaceous plants ($n = 12$). Similarly no cover differences were found over all sites for total herbaceous species, grasses, or forbs. We rejected our hypotheses for shrubs as elk herbivory did negatively impact shrub recovery after wildfire. Reductions of shrub cover and productivity in mountain big sagebrush-grass communities from wildfire and intense herbivory have implications for many organisms. Keywords: *Artemisia tridentata*, elk, fire, sagebrush, Yellowstone

INTRODUCTION

Ungulates rely on sagebrush (*Artemisia*) habitat types for winter forage on the Northern Yellowstone Winter Range (NYWR) (Wambolt 1996; Wambolt 1998; Wambolt and McNeal 1987; Wambolt and Sherwood 1999). Basin big sagebrush (*A. t. tridentata*) and Wyoming big sagebrush (*A. t. wyomingensis*), are important on the NYWR. However, it is mountain big sagebrush (*A. t. vaseyana*) that is most important there due to the larger area it occupies and its high preference by ungulates as forage (Wambolt 1996; Wambolt 1998).

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Carl L. Wambolt, Professor of Range Science, Department of Animal and Range Sciences, Montana State University, Bozeman.

²Reyer J. Rens, former graduate assistant, currently with Bureau of Land Management, Dillon, MT

During winter basin big sagebrush is particularly important for ungulates as nutritious forage and for thermal and security cover (Wambolt 2004; Welch and McArthur 1979). Mechanisms that influence ungulate browsing patterns on sagebrush taxa on the NYWR have been investigated, including intensity of use (Bray and others 1991; Personius and others 1987; Singer and Renkin 1995; Striby and others 1987; Wambolt 1996; Wambolt and McNeal 1987; Wambolt and others 1994). Intense browsing has reduced big sagebrush populations on the NYRW (Wambolt 1996; Wambolt and others 1999; Wambolt and Sherwood 1999). Wildfire at least temporarily eliminates big sagebrush taxa as they respond negatively to fire (Wambolt and others 1999; Welch 2005). Loss of big sagebrush can impact numerous wildlife populations (Welch 2005).

During the early twentieth century, concern for the welfare of big sagebrush communities within Yellowstone National Park (YNP) received much attention (Cahalane 1943; Kittams 1950; Rush 1932; Wright and Thompson 1935). In response, the National Park Service (NPS) constructed 10 exclosures in 1957 and 1962, partially to study the relationships between ungulate foraging and big sagebrush. Two exclosures on the Black-tailed Deer Plateau containing only big sagebrush-grass habitat types completely burned during the large Yellowstone fires of 1988 creating an opportunity to investigate fire and elk herbivory influences on these communities.

Our objective was to investigate recovery of mountain big sagebrush and associated species on the Black-tailed Deer Plateau of the NYWR a decade after the 1988 Yellowstone wildfires, a period during which the area was under relatively intense winter herbivory, largely by elk. We did this by comparing shrub and herbaceous cover and shrub densities between environmentally paired sites protected or unprotected from ungulate foraging within the area burned in 1988. We tested the hypotheses that these plant taxa recovered from the wildfire equally with or without elk herbivory.

STUDY SITES

The NYWR occupies 100,000 ha over an 80 km stretch along the lower elevations in northern YNP and extends northward into Montana along the Yellowstone, Gardner and Lamar drainages (Houston 1982). The Black-tailed Deer Plateau where the study was conducted is approximately midway between Mammoth and Tower Junction and near the center of the NYWR. The NPS

erected two 2 ha ungulate exclosures on this plateau, one in 1957 and the second in 1962, to evaluate the effects of ungulate grazing on mountain big sagebrush communities (Barmore 1980; Houston 1982; Singer and Harter 1996). The elevation is approximately 2,040 m, midway between the lower (1,600 m) and higher (2,600 m) elevations of the NYWR (Coughenour 1991; Houston 1982). Annual precipitation received at Mammoth (1,899 m) is approximately 400 mm, and at Tower Falls (1,912 m) 410 mm, the nearest weather stations (Farnes 1991). Half of the precipitation is received as snow, and most rainfall occurs in spring and early summer with the peak received in June (50mm) (Farnes 1991). The decade following the 1988 wildfires in YNP was characterized by winters of less than average severity (Farnes 1991). Pinedale glacial till is the parent material for Mollisols on the study area with a cryic temperature regime (Lane 1990). Texture is typically loam with coarse fragments ranging from gravel to boulders (Lane 1990). The mountain big sagebrush and bluebunch wheatgrass (*Agropyron spicatum*) habitat type dominates, especially on slopes with any south aspect. The mountain big sagebrush and Idaho fescue (*Festuca idahoensis*) habitat type is more likely found on north and east facing slopes. Green rabbitbrush (*Chrysothamnus viscidiflorus*), rubber rabbitbrush (*C. nauseosus*), and gray horsebrush (*Tetradymia canescens*) are less dominant than mountain big sagebrush, but common. These three shrubs have a sprouting habit.

METHODS

Sampling was conducted during the summers of 1998 and 1999, a decade after the wildfire. Twelve environmentally different sites based on slope and aspect were found in the two exclosures (Coughenour 1991). These 12 sites were paired with adjacent or nearby unprotected sites (table 1). Mountain big sagebrush, sprouting shrubs (green rabbitbrush, rubber rabbitbrush, and gray horsebrush), and herbaceous species dominated five, four, and three sites, respectively. However, all 12 sites were used in comparing cover of herbaceous species because they were important on all sites. Only established shrubs were sampled for density to avoid occasional abnormally high numbers of seedlings that always have a high mortality. Established shrubs were those with an average cover ≥ 15 cm from four intercepts (Wambolt and others 1994).

At each of the 12 sites, 10 stratified 30 m transects were located inside the exclosure as were 10 transects outside the exclosure on an environmentally paired site (Coughenour 1991; Wambolt and Sherwood 1999). The line interception method (Canfield 1941; Wambolt and others 2006) was

used to obtain shrub cover. All shrub cover segments ≥ 3 cm were included. Open spaces and dead foliage ≥ 3 cm were excluded. A plumb bob was projected from the transect line to plants to determine precise intercepts. Along each 30 m line, belt transects of 2 x 30 m were created by measuring 1 m on both sides of the line transect. Densities of established shrubs were obtained within the 10 belt transects. A decimeter (20 x 50 cm) frame was placed at every third meter along each transect to estimate cover of herbaceous species (Daubenmire 1959). This resulted in 100 frames in each the protected and browsed-grazed portions of each environmental pairing.

Table 1—Slopes and aspects at the 12 locations where browsed and/or grazed sites and protected sites were paired by both characters. The paired sites are associated with two exclosures established in 1957 and 1962 on the Black-tailed Deer Plateau and burned during the Yellowstone fires of 1988. Sampling was conducted during the summers of 1998 and 1999.

Site number ^a	Slope ^d (%)	Aspect (°from north)
1	Moderate	132
2	Steep	160
3 ^b	Gentle	165
4 ^b	Flat	
5	Moderate	122
6 ^c	Gentle	82
7 ^b	Gentle	180
8 ^c	Moderate	144
9 ^c	Moderate	138
10 ^c	Moderate	154
11 ^c	Steep	116
12 ^b	Gentle	71

^aAll sites were used in herbaceous species comparisons.

^bSites used in sprouting shrub comparisons.

^cSites used in mountain big sagebrush comparisons.

^dSlope classes are: Flat ≤ 3 percent, Gentle = 4–15 percent, Moderate = 16–29 percent, Steep = 30–44 percent.

Two sample t procedures were used to compare cover (shrub and herbaceous) and density (shrub) means between protected and browsed-grazed sites after insuring the data were normally distributed (Coughenour 1991; Wambolt and Sherwood 1999). To make simultaneous inferences at the $P = 0.05$ level, the tests were adjusted using the Bonferroni procedure (Neter and others 1996).

RESULTS

The comparison between protected and browsed areas over all five mountain big sagebrush dominated sites found mountain big sagebrush cover lower ($P \leq 0.001$) on browsed areas at 9.7 percent compared to 20 percent where protected (table 2). The three sprouting shrubs collectively

comprised considerably less cover than mountain big sagebrush (table 2). However, relationships were similar. Sprouting shrubs had less cover ($P \leq 0.001$) at two percent on browsed areas than on unbrowsed areas at 5.8 percent on the four sites on which they were the dominant shrubs (table 2).

Table 2—Percent cover ($n = 10$) of all mountain big sagebrush and sprouting shrubs (Green rabbitbrush, rubber rabbitbrush, and gray horsebrush) with standard deviations in parentheses at environmentally paired protected or browsed sites on the Black-tailed Deer Plateau of Yellowstone National Park and sampled the summers of 1998 and 1999.

Site	Protected (%)	Browsed (%)	Probability $> t^a$
Mountain big sagebrush			
6	16.3 (5.2)	7.9 (5.2)	0.002
8	20.3 (5.5)	6.3 (4.4)	0.001
9	24.8 (6.4)	10.9 (3.8)	0.001
10	18.1 (8.2)	11.4 (5.5)	0.045
11	20.5 (5.0)	11.9 (3.1)	0.001
Sprouting shrubs			
3	4.3 (2.3)	0.7 (0.4)	0.001
4	4.4 (2.6)	2.9 (1.5)	0.120
7	6.7 (3.4)	0.9 (0.6)	0.001
12	7.6 (3.4)	3.7 (1.6)	0.006

^aThe comparative P value for similar inferences is 0.01.

Over the five mountain big sagebrush sites density was not reduced ($P \leq 0.12$) by browsing. Established mountain big sagebrush densities ranged from 45.9 to 147.8 and 32.5 to 129.4 per 60 m² where protected and browsed, respectively (table 3). Individual comparisons of four of the five sites dominated by mountain big sagebrush showed no differences ($P \leq 0.05$) in density, although site 11 had more plants ($P \leq 0.018$) where protected (87.6 per 60 m²) than where browsed (71.4 per 60 m²). The density of sprouting shrubs was not different ($P \leq 0.47$) across the four sites they dominated at an average of 73.9 and 82.2 per 60 m² where protected and browsed, respectively (table 3).

Over all 12 sites total herbaceous cover was not different ($P \leq 0.37$) at 51.3 percent and 52.4 percent where protected and grazed, respectively (table 4). Grasses comprised the majority of the herbaceous cover with averages of 36.7 percent and 38.6 percent where protected and grazed, respectively ($P \leq 0.085$). Forb cover was not different ($P \leq 0.27$) either with averages of 14.6 percent and 13.9 percent where protected and grazed, respectively.

DISCUSSION

Shrub Canopy Cover

Our results indicate browsing suppressed recovery of shrubs after fire. Browsing lessened mountain big sagebrush dominance when it resulted in a decline of cover

from 20 percent to 9.7 percent ($P \leq 0.001$) on our sites. On unburned portions of the NYWR, Wambolt and Sherwood (1999) reported similar results from communities established much longer with no known fire effects since YNP was created in 1872. Their protected sites had not been browsed for either 32 or 37 years. They found average big sagebrush cover on protected portions was three times greater ($P \leq 0.0027$) than where browsed over 19 sites. Their cover averaged 19.7 percent inside and 6.5 percent outside at 11 mountain big sagebrush and eight Wyoming big sagebrush sites. Blaisdell (1953) reported little re-establishment of mountain big sagebrush 12 years after burning in Idaho. In a similar manner, just north of YNP on seven NYWR sites burned 10 to 14 years previously, mountain big sagebrush cover averaged 1.2 percent compared to 33 unburned sites that averaged 14.2 percent despite very heavy browsing (Wambolt and others 1999).

Table 3—Density of established mountain big sagebrush plants and sprouting shrubs (Green rabbitbrush, rubber rabbitbrush, and gray horsebrush) per 60 m² ($n = 10$) with standard deviations in parentheses at environmentally paired protected or browsed sites on the Black-tailed Deer Plateau of Yellowstone National Park and sampled during the summers of 1998 and 1999.

Site	Protected (shrubs per 60 m ²)	Browsed (shrubs per 60 m ²)	Probability $> t^a$
Mountain big sagebrush			
6	45.9 (19.1)	32.5 (11.1)	0.071
8	74.4 (16.3)	42.7 (44.8)	0.059
9	147.8 (49.0)	129.4 (58.4)	0.458
10	99.1 (27.5)	103.9 (33.7)	0.731
11	87.6 (15.4)	71.4 (12.2)	0.018
Sprouting shrubs			
3	67.0 (12.9)	43.3 (17.8)	0.003
4	89.3 (37.1)	88.1 (24.3)	0.933
7	63.3 (17.8)	30.1 (14.2)	0.001
12	76.2 (21.4)	167.5 (73.2)	0.003

^aThe comparative P value for similar inferences is 0.01.

Ordinarily the sprouting shrubs we sampled should be expected to flourish after fire with light to moderate browsing (Wambolt and others 1999; Wambolt and Sherwood 1999), until successional replaced by the habitat type dominant, mountain big sagebrush (Blaisdell 1953; Wambolt and Payne 1986; Young and Evans 1974). However, we found 2 to 3 times as much sprouting shrub cover on unbrowsed sites as compared to browsed sites a decade after fire. Wambolt and Sherwood (1999) reported two percent canopy cover of sprouting shrubs in protected areas and 1.1 percent where browsed ($P \leq 0.0027$) on 19 unburned areas across the NYWR. Mehus (1995) working on the NYWR, but north of YNP also found cover of *Chrysothamnus* taxa greater ($P \leq 0.001$) where burned (8.5 percent) 19 years earlier than on unburned areas

(0.9 percent). These earlier studies support our findings that intense herbivory has reduced sprouting shrubs, despite their relative resiliency to browsing.

Table 4—Cover ($n = 100$) of all herbaceous vegetation and perennial grasses with standard deviations in parentheses at protected or grazed sites on the Black-tailed Deer Plateau of Yellowstone National Park that were environmentally paired and sampled the summers of 1998 and 1999.

Site	Protected (%)	Grazed (%)	Probability > t^a
Herbaceous vegetation			
1	56.8 (7.8)	58.9 (9.6)	0.516
2	37.3 (8.3)	45.5 (6.4)	0.022
3	49.3 (5.1)	51.2 (3.7)	0.358
4	58.6 (5.6)	54.0 (3.3)	0.040
5	64.5 (8.7)	55.3 (3.4)	0.022
6	62.5 (7.2)	56.2 (4.3)	0.029
7	52.3 (3.2)	42.5 (3.4)	0.001
8	52.6 (6.8)	59.8 (4.7)	0.017
9	39.2 (6.4)	42.9 (6.7)	0.221
10	41.6 (10.3)	50.9 (6.0)	0.024
11	48.6 (6.3)	56.9 (6.1)	0.008
12	55.2 (3.0)	55.9 (3.9)	0.636
Perennial grasses			
1	50.9 (6.3)	44.1 (4.3)	0.028
2	23.1 (7.2)	31.3 (5.9)	0.012
3	38.1 (5.3)	35.3 (4.8)	0.226
4	42.0 (6.7)	39.4 (2.9)	0.288
5	32.9 (5.0)	42.3 (4.2)	0.001
6	49.4 (6.1)	41.9 (4.9)	0.007
7	37.7 (2.8)	30.8 (4.0)	0.001
8	38.2 (5.5)	46.7 (6.0)	0.005
9	27.2 (5.6)	32.3 (5.8)	0.062
10	28.9 (8.6)	36.4 (3.5)	0.027
11	32.3 (3.2)	42.5 (3.1)	0.001
12	39.8 (2.8)	40.9 (2.4)	0.334

^aThe comparative P value for similar inferences is 0.0042.

Shrub Density

Mountain big sagebrush establishment and thereby density was not affected ($P \leq 0.12$) by elk browsing in the significant manner that growth and subsequently cover was in the decade after the 1988 fire (table 3). This may be partially explained by the fact that sites nine and 10 are located in depressions that accumulate snow, thereby protecting small establishing plants from browsing. Wambolt and Sherwood (1999) determined browsing on 19 unburned NYWR sites was intense enough to reduce sagebrush density. They found 30.5 plants where browsed and 15.3 plants (per 60 m²) ($P \leq 0.0027$) where protected 32 or 37 years with no known fire events since establishment of YNP. Our results likely differ due to the interaction of

browsing and fire over a shorter period of one decade with winters of less than average severity (Farnes 1991). On intensively browsed, burned and unburned portions of the NYWR north of YNP, Wambolt and others (1999) found an average of 52 plants on 33 unburned sites and 14 plants on seven burned sites, per 60 m², 10 to 14 years after fire. Wambolt and Hoffman (2001) reported that 47 percent of the mountain big sagebrush plants that established between 1978 and 1992 did so during 1988 due to a series of ideal conditions. We believe that if these ideal conditions for shrub establishment had not coincided with the fire, that both the mountain big sagebrush density and cover we found would be considerably lower.

Our findings (table 3) regarding densities of sprouting shrubs are quite variable among sites. This is likely due to the southerly exposures of sites three and seven accumulating less snow and therefore, providing more browsing opportunities for elk than on sites four and 12. Our results for sprouting shrubs on our burned study area again differ from unburned portions of the NYWR (Wambolt and Sherwood 1999). While our sprouting shrub densities were not always less where browsed, Wambolt and Sherwood (1999) found consistently higher densities ($P \leq 0.0027$) where protected (16.6 per 60 m²) than on browsed portions (13.1 per 60 m²) of 19 sites. The higher densities (table 3) we found following burning are very common without intense browsing (Blaisdell 1953; Wambolt and others 1999; Wambolt and Payne 1986; Young and Evans 1974). Mehus (1995) on the NYWR north of YNP found nine times as many rabbitbrush plants on the portion of a site burned 19 years earlier compared to the unburned portion.

Evidently mountain big sagebrush in and out of the exclosures had an equal opportunity to reestablish. Big sagebrush reestablishment after fire is highly dependent on survival of mature plants to serve as a seed source (Johnson and Payne 1968). However, the 1988 fire burned intensely enough to eliminate all mature mountain big sagebrush plants on our study area. The even-aged distribution of mountain big sagebrush plants across the area indicates reestablishment occurred from soil born seed surviving the fire and germinating that growing season (Wambolt and others 1999). These seedlings became established through a series of ideal weather conditions and a 40 percent reduction in elk numbers (Wambolt and others 1999).

Herbaceous Plants

We found little impact from winter grazing on the herbaceous species. Cover for the dominant herbaceous species, *Agropyron spicatum* and *Festuca idahoensis*, was not different where grazed or protected. The same was true for forbs over all 12 study sites. Others have reported little difference in grass or forb production on the NYWR from either fire or grazing compared to protected areas

(Coughenour 1991; Singer and Harter 1996). Similarly without considering the interaction with fire that our study included, little change in herbaceous parameters due to herbivory has been found on the NYWR (Coughenour 1991; Houston 1982). Understory cover was not lower on sites with higher shrub cover. This has been found in widespread previous research which indicated the absence of increased herbaceous production from sagebrush reduction (Anderson and Holte 1981; Blaisdell 1953; Daubenmire 1975; Kuntz 1982; Peek and others 1979; Wambolt and others 2001). Because our measurements were made 10 years post-fire, grazing impacts may have been smaller than if measured within a couple of years after the fire (Singer and Harter 1996). However, nine years after another YNP fire Houston (1982) did find nearly twice as much production from perennial grasses on an unburned portion of range compared to a burned portion with 138 g per m² and 79 g per m², respectively.

CONCLUSIONS

Herbaceous plants are known to generally sustain non-growing season grazing without negative effects (Cook and Stoddart 1963). In addition, herbaceous plants are often relatively unavailable for intense winter grazing due to snow cover. There were no important effects from elk winter grazing on herbaceous species following the 1988 wildfire on our study area. Thus, we accept our hypothesis for herbaceous plants, that elk herbivory did not influence their recovery from fire.

However, the impact of ungulate winter browsing on unburned established stands of mountain big sagebrush on the NYWR is documented to include declines (Mehus 1995; Wambolt 1998; Wambolt and Sherwood 1999). Investigations in the greater Yellowstone area of fire impacts on this important NYWR taxon indicate that typically there is minimal recovery of mountain big sagebrush stands for several decades following fire (Wambolt 1998; Wambolt and McNeal 1987; Wambolt and others 1999; Wambolt and others 2001). Fortunately for sagebrush dependant wildlife, 1988, the year of the large Yellowstone fires provided ideal conditions for nearly half of all mountain big sagebrush established on the NYWR between 1978 and 1992 to begin growth (Wambolt and Hoffman 2001). Relatively rapid establishment was promoted by favorable germination and establishment conditions the year following the fire. Without these favorable conditions, undoubtedly mountain big sagebrush cover and density would have been much lower only a decade after the fire when we conducted our investigation. Thus, it is not surprising we found mountain big sagebrush

plants did establish equally on sites either protected or available to elk browsing a decade after fire.

In considering the ecological implications of NYWR browsing on sagebrush communities elsewhere it is important to realize that our study sites had not been subjected to as much past herbivory as were other NYWR sites previously documented that typically have more open winters for foraging (Mehus 1995; Wambolt 1998; Wambolt and others 1999; Wambolt and Sherwood 1999). This was partially due to our study site location which was determined by the earlier placement of the exclosures and subsequently the 1988 fire. Also important was the less than average winter severity during the decade after the wildfires on the NYWR (Farnes 1991).

Most importantly we learned that regardless of past browsing levels and fast recovery from the fire regardless of browsing, that browsing affects the growth of establishing mountain big sagebrush plants. The cover of the mountain big sagebrush plants was significantly reduced from browsing, thereby reducing the amount of forage they produce for wintering ungulates like elk as well as other habitat values (Wambolt and others 1994). We thereby reject our hypothesis regarding mountain big sagebrush and conclude that elk herbivory negatively impacted the taxon's recovery after this wildfire.

Green rabbitbrush, rubber rabbitbrush, and gray horsebrush, all shrubs commonly associated with mountain big sagebrush were impacted similarly. Their decline from fire and browsing is typically unexpected, partially because all of these shrubs have a habit of sprouting when disturbed that generally results in increased growth, not a decline (Wambolt and Sherwood 1999). A decade after the fire the sprouting shrubs show the same herbivory diminished recovery pattern as mountain big sagebrush indicating an unusually heavy browsing effect. Therefore, we also reject our hypothesis regarding the sprouting shrubs collectively.

Although this study considered plant recovery after wildfire with intense elk herbivory, our plant response findings parallel previous findings on the NYWR that did not include fire effects. We conclude the impact of elk foraging was the same during the decade after the 1988 fire as found previously and greatly reduced the shrubs of mountain big sagebrush communities. The decline of all the shrub taxa on mountain big sagebrush-grass range has the potential to negatively affect numerous wildlife species that rely on the shrubs as forage or security-thermal cover (Welch 2005).

REFERENCES

- Anderson, J.E., Holte, K.E. 1981. Vegetation development over 25 years without grazing on sagebrush-dominated rangeland in southeastern Idaho. *Journal of Range Management*. 34:25–29.
- Barmore, W.J. 1980. Population characteristics, distribution and habitat relationships of six ungulates in northern Yellowstone National Park. Final Report. Yellowstone National Park files.
- Blaisdell, J.P. 1953. Ecological effects of planned burning of sagebrush-grass range on the upper Snake River Plains. USDA Technical Bulletin 1075. Ogden, UT.
- Bray, R.O., Wambolt, C.L., Kelsey, R.G. 1991. Influence of sagebrush terpenoids on mule deer preference. *Journal of Chemical Ecology*. 17:2053–2062.
- Cahalane, V.H. 1943. Elk management and herd regulation – Yellowstone National Park. Transactions of the North American Wildlife Conference. 8:95–101.
- Canfield, R. 1941. Application of the line interception method on sampling range vegetation. *Journal of Forestry*. 39:388–394.
- Cook, C.W., Stoddart, L.A. 1963. The effect of intensity and season of use on the vigor of desert range plants. *Journal of Range Management*. 16:315–317.
- Coughenour, M.B. 1991. Biomass and nitrogen responses to grazing of upland steppe on Yellowstone's northern winter range. *Journal of Applied Ecology*. 28:71–82.
- Daubenmire, R. 1959. A canopy-coverage method of vegetational analysis. *Northwest Science*. 33:43–64.
- Daubenmire, R. 1975. Plant succession on abandoned fields and fire influences in a steppe area in southeastern Washington. *Northwest Science*. 49:36–48.
- Farnes, P.E. 1991. A scaled index of winter severity. Pages 135–138 in Proceedings of the Western Snow Conference. 12–15 April 1991, Juneau, Alaska, Colorado State University Press, Fort Collins.
- Houston, D.B. 1982. The northern Yellowstone elk, ecology and management. Macmillan Publishers, New York, New York.
- Johnson, J.R., Payne, G.F. 1968. Sagebrush reinvasion as affected by some environmental influences. *Journal of Range Management*. 21:209–213.
- Kittams, W.H. 1950. Sagebrush on the lower Yellowstone range as an indicator of wildlife stocking. Yellowstone National Park files.
- Kuntz, D.E. 1982. Plant response following spring prescribed burning in an *Artemisia tridentata* ssp. *vaseyana*-*Festuca idahoensis* habitat type. Doctor of Philosophy Dissertation, University of Idaho, Moscow.
- Lane, J. 1990. Characterization and comparison of soils inside and outside of grazing exclosures on Yellowstone National Park's northern winter range. Master of Science Thesis, Montana State University, Bozeman.
- Mehus, C.A. 1995. Influences of browsing and fire on sagebrush taxa on the Northern Yellowstone Winter Range. Master of Science Thesis, Montana State University, Bozeman.
- Neter, J., Kutner, M.H., Nachtsheim, C.J., Wasserman, W. 1996. Applied Linear Statistical Models. Irwin Publishing. 4th Edition.
- Peek, J.M., Riggs, R.A., Lauer, J.L. 1979. Evaluation of fall burning on bighorn sheep winter range. *Journal of Range Management*. 32:430–432.
- Personius, T.L., Wambolt, C.L., Stephens, J.R., Kelsey, R.G. 1987. Crude terpenoid influence on mule deer preference for sagebrush. *Journal of Range Management*. 40:84–88.
- Rush, W.M. 1932. Northern Yellowstone Elk Study. Montana Fish and Game Commission, Helena.
- Singer, F.J., Harter, M.K. 1996. Comparative effects of elk herbivory and 1988 fires on northern Yellowstone National Park grasslands. *Ecological Applications*. 6:185–199.
- Singer, F.F., Renkin, R.A. 1995. Effects of browsing by native ungulates on the shrubs in big sagebrush communities in Yellowstone National Park. *Great Basin Naturalist*. 55:201–212.
- Striby, K.D., Wambolt, C.L., Kelsey, R.G., Havstad, K.M. 1987. Crude terpenoid influence on in vitro digestibility of sagebrush. *Journal of Range Management*. 47:231–234.
- Wambolt C.L. 1996. Mule deer and elk foraging preference for 4 sagebrush taxa. *Journal of Range Management*. 49:499–503.
- Wambolt, C.L. 1998. Sagebrush and ungulate relationships on Yellowstone's northern range. *Wildlife Society Bulletin*. 26:429–437.
- Wambolt, C.L. 2004. Browsing and plant age relationships to winter protein and fiber of big sagebrush subspecies. *Journal of Range Management*. 57:620–623.
- Wambolt, C.L., Creamer, W.H., Rossi, R.J. 1994. Predicting big sagebrush winter forage by subspecies and browse form class. *Journal of Range Management*. 47:231–234.
- Wambolt, C.L., Frisina, M.R., Knapp, S.J., Frisina, R.M. 2006. Effect of method, site, and taxon on line intercept estimates of Sagebrush cover. *Wildlife Society Bulletin*. 34:440–445.
- Wambolt, C.L., Hoffman, T. 2001. Mountain big sagebrush age distribution and relationships on the northern Yellowstone winter range. Pages 330–335 in: E.D. McArthur and D.J. Fairbanks, compilers, Proceedings of the symposium, Shrubland ecosystem genetics and biodiversity. USDA Forest Service Proceedings RMRS-P-21. Ogden, UT.
- Wambolt, C.L., Hoffman, T.L., Mehus, C.A. 1999. Response of shrubs in big sagebrush habitats to fire on the Northern Yellowstone Winter Range. Pages 238–242 in: E.D. McArthur, W.K. Ostler, C.L. Wambolt, compilers, Proceedings of the symposium, Shrubland ecotones. USDA Forest Service Proceedings. RMRS-P-11. Fort Collins, CO.
- Wambolt, C.L., McNeal, A.F. 1987. Selection of winter foraging sites by elk and mule deer. *Journal of Environmental Management*. 25:285–291.
- Wambolt, C.L., Payne, G.F. 1986. An 18-year comparison of control methods for Wyoming big sagebrush in Southwestern Montana. *Journal of Range Management*. 39:314–319.
- Wambolt, C.L., Sherwood, H.W. 1999. Sagebrush response to ungulate browsing in Yellowstone. *Journal of Range Management*. 52:363–369.
- Wambolt, C.L., Walhof, K.S., Frisina, M.R. 2001. Recovery of big sagebrush communities after burning in southwestern Montana. *Journal of Environmental Management*. 61:323–348.
- Welch, B.L. 2005. Big sagebrush: a sea fragmented into lakes, ponds and puddles. United States Forest Service, General Technical Report RMRS-GTR-144.
- Welch, B.L., McArthur, E.D. 1979. Feasibility of improving big sagebrush for use on mule deer winter ranges. Pages 451–457 in: J.R. Goodin, and D.K. Northington editors, Arid plant resources. Texas Tech University, Lubbock.
- Wright, G.M., Thompson, B.H. 1935. Fauna of the National Parks of the United States: wildlife management in the national parks. National Park Service Fauna Series 2.
- Young, J.A., Evans, R.A. 1974. Population dynamics of green rabbitbrush in disturbed big sagebrush communities. *Journal of Range Management*. 27:127–132.

Post-fire Recovery of Wyoming Big Sagebrush Steppe in Central and Southeast Montana

Stephen V. Cooper¹, Peter Lesica², and Greg M. Kudray³

ABSTRACT

Big sagebrush (*Artemisia tridentata* Nutt.) steppe is a widespread habitat throughout eastern Montana and supports several sagebrush-dependent species including Greater Sage-grouse (*Centrocercus urophasianus*). We sampled 24 burned-unburned paired sites in central and southeastern Montana dominated by Wyoming big sagebrush (ssp. *wyomingensis*). Time since fire ranged from 4 to 67 years. Prescribed burns and wildfires typically resulted in the complete mortality of big sagebrush. We found negligible post-fire Wyoming big sagebrush recovery for 17 of the 24 sites and the oldest burn (67 years) was only 8 percent recovered. Perennial grass cover increased 27 percent and 20 percent following prescribed fire and wildfire, respectively; western wheatgrass (*Pascopyrum smithii*) accounted for most of this increase. Annual grass cover increased by 11 percent due primarily to field brome (*Bromus arvensis*, formerly *B. japonicus*). Plant species richness significantly declined in burned plots compared to unburned controls. There was no change after burning in overall sub-shrub or forb cover or the density of *Cichorieae* forbs that are important for successful Greater Sage-grouse brood rearing. Managers concerned about Greater Sage-grouse and other sage-dependent species should be extremely cautious with prescribed burns and wildfires in this region. Fire will likely eliminate sagebrush habitat, increase weedy annual grass cover and reduce species richness; sagebrush cover could take a century or more to recover to pre-burn conditions.

INTRODUCTION

Sagebrush steppe is a dominant vegetation type in the Great Basin and Intermountain Region of western North America but it is also important in portions of the Northern Great Plains where agriculture (cereal grains) and mixed-grass prairie now dominate. Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis* Beetle and Young) - dominated vegetation is an important component of the semiarid landscapes east of the Rocky Mountains stretching from Wyoming through Montana and western North Dakota to just south of the Canadian border. Physiognomy of Wyoming big sagebrush stands in the Northern Great Plains differs from that of the Intermountain Region by having undergrowth generally dominated by rhizomatous grasses as opposed to tussock-forming grasses. Also influencing stand physiognomy are two notable clines in

Wyoming big sagebrush size presumed to reflect available soil moisture; one of increasing plant height from 1) south to north and 2) from lower to higher elevation.

Fire was instrumental in structuring presettlement sagebrush ecosystems, generating a mosaic of stands of different size and various seral stages (West 2000). Conserving native species diversity is expected to require maintaining a comparable mosaic. Greater Sage-grouse (*Centrocercus urophasianus*), for example, require barren habitats for leks, relatively dense stands of medium height for nesting (Klebenow 1969; Wallestad and Pyrah 1974; Aldridge and Brigham 2002), open stands for brood raising (Klebenow 1973; Wallestad 1971), and full-canopied tall stands for wintering (Eng and Schladweiler 1972). Greater Sage-grouse populations apparently can be constrained by the loss of any one of these structural types (Connelly and others 2000; Roscoe 2002). Antelope, Brewer's Sparrow, Sage Sparrow, Sage Thrasher and Sagebrush Vole are also sagebrush dependent at some stage of their life cycles.

Management strategies that promote the conservation of all sagebrush steppe-dependent species are currently being formulated, and prescribed fire has been proposed as a method to control the density of big sagebrush stands (Klebenow 1973; Pyle and Crawford 1996). However, we only have limited knowledge of changes in sagebrush cover, height, associated vegetation and other characteristics as succession proceeds from immediate post-treatment to mature structure. Though considered a climax-dominant species, evidence suggests that big sagebrush fire response varies according to subspecies and may require many years for post-fire re-establishment (Baker 2007). Wyoming big sagebrush, although highly variable in response (Walhof 1997; Wambolt and others 2001; Watts and Wambolt 1996), demonstrates little recovery for the first 30 years (Wambolt and Payne 1986; Eichhorn and Watts 1984; Lesica and others 2007) and generally requires at least 50 years to attain a density equal to that of the unburned control (Baker 2007; Colket 2003); a state-and-transition model developed for ssp. *wyomingensis* within the Columbia Basin is distinctly more optimistic envisioning recovery in approximately 30 years (Hemstrom and others 2002). With the lone exception of the Eichhorn and Watts (1984) study in central Montana's Missouri River Breaks, none of these studies were conducted in a Great Plains environment. The ecological dynamics and habitat characteristics of these sagebrush communities are

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Stephen V. Cooper, Montana Natural Heritage Program (Retired), Helena, MT

²Peter Lesica, Conservation Biology Research, Missoula, MT

³Greg M. Kudray, Montana Natural Heritage Program, Helena.

almost certainly strongly influenced by their age (size) structure. Landscape scale comprehensive management of sagebrush cannot be achieved without understanding how structural and compositional components change with time since disturbance.

We also have little knowledge of how undergrowth vegetation changes with succession. Fire often increases the abundance of grasses (Harness and Murray 1973; Wambolt and Payne 1986). Forbs, especially those in the *Cichorieae* Tribe of the *Asteraceae*, are an important component of sage grouse diet during the summer and are often critical for successful brood rearing (Klebenow and Gray 1967; Peterson 1970; Barnett and Crawford 1994; Drut and others 1994). At what age do stands provide the most succulent forbs for sage grouse brood rearing, the best nesting cover for grouse, the best habitat for Brewer's sparrows and sage thrashers? Fire is reported to have positive effects on grass productivity in other regions of the western U.S. (Pechanec and others 1954; Harniss and Murray 1973; Bunting and others 1998), but there is little information on whether this is true for the Great Plains.

The purpose of this study was to describe and model the change in sagebrush and associated vegetation after fire in the Northern Great Plains of eastern and central Montana. We documented changes in shrub height, cover and size-class distribution and canopy cover of undergrowth herbaceous species by sampling burned and adjacent unburned stands of various post-fire ages.

STUDY AREA

Sampling was conducted over a broad swath of eastern Montana from a westernmost site within the Bighorn Basin to the glaciated plains north of the Missouri River. Most of the sampling occurred in the Tongue and Powder River Basins and in the Missouri River Breaks (figure 1). Elevations ranged from 270 to 1,220 m (890 to 3,990 ft). Annual precipitation varied from 274 mm (10.8 in) in the Missouri River Breaks to 415 mm (16.35 in), on plateaus east of the Powder River (DAYMET model, Thornton and others 1997). Approximately 45 percent of this precipitation occurs in the biologically critical spring quarter (April-June). The continental climate is characterized by strong seasonal (winter to summer) and diurnal temperature fluctuations. All the sampling sites were characterized as flat to gently rolling with no slopes greater than 13 percent and were post-sampling keyed (based on laboratory determination of soil texture) to one of three Natural Resource Conservation Service (NRCS) "Ecological Site" types; Silty (Si, 58 percent of sites); Clayey (Cy, 32 percent); Sandy (Sy, 10 percent). Soil texture ranged from clays to sandy loam with no true "sands" type being represented.

Wyoming big sagebrush was the only big sagebrush subspecies at sampling sites, although silver sagebrush (*A. cana* ssp. *cana*) occurred on nearby stream terraces and sites having a greater percentage of sand in the soil. All the sites were characterized as flat to gently rolling with no slopes greater than 13 percent. Soils texture ranged from clays to sandy loam. Dominant undergrowth grasses were bluebunch wheatgrass (*Pseudoroegneria spicata*) on the driest sites to Idaho fescue (*Festuca idahoensis*) on the most mesic sites; the most common native species were western wheatgrass (*Pascopyrum smithii*), green needlegrass (*Nassella viridula*), Sandberg's bluegrass (*Poa secunda*), blue grama (*Bouteloua gracilis*), and needle-and-thread (*Hesperostipa comata*).

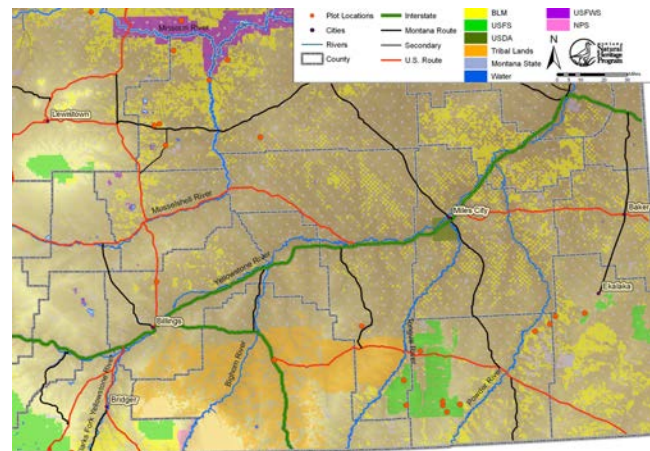


Figure 1—Study area map with sampling sites (plots) indicated.

METHODS

Field Methods

In June and July of 2006 and 2007 we sampled 24 sites dominated by Wyoming big sagebrush in central and southeastern Montana. We used lists of potential sites provided by the Miles City Office of the Bureau of Land Management (BLM), the Ashland Ranger District of the Custer National Forest, and personal communication with their respective staffs to select sampling sites based on age of burn and accessibility. Larry Eichhorn, retired BLM range conservationist from Lewistown, provided information valuable in relocating the original sample sites of his study of post-fire succession in central Montana (Eichhorn and Watts 1984). We focused on federally or state owned lands but fortuitously encountered several privately held sites with cooperative owners.

At each site a 1000 m² macroplot (20 m by 50 m) was visually selected to represent prevailing conditions within the burned area and be as close as feasible and preferably on the downwind side of the unburned area. A "control" sample macroplot was established in unburned sagebrush-

dominated vegetation as close as possible to the burn. The control was chosen to be as similar as possible to the abiotic setting (slope, aspect, soils) of the burned sample plot. Although the unburned control macroplots are not true controls because of not being randomly assigned prior the fires, nonetheless they function as controls by exemplifying what the burned plot probably would have constituted, had they not burned. With one exception, burned macroplots were located within 25 m or less of the unburned control and always in the same grazing pasture (not separated by fencing). We noted the positions of fence lines and water developments and attempted to locate sampling points as far removed as possible to ensure that grazing pressure was not excessive. However, we had no way of accurately accounting for grazing regimes.

We used line intercept to estimate shrub canopy cover along five evenly-spaced, parallel 20 m transects originating at the 50 m macroplot baseline (Mueller-Dombois and Ellenberg 1974). Circular microplots 3m² were established at the 5 and 10 m marks of the five transect lines (10 total) for determining rooted density for all shrub species by four size classes: (1) seedlings (< 10 cm high), (2) juveniles (> 10 cm and stem diameter at ground level < 1 cm), (3) sub-adults (stem 1 to 3 cm diameter), and (4) adults (> 3 cm stem diameter). Age and height were recorded for one sagebrush plant of each size class in alternate microplots (5 total), focusing on specimens exhibiting the least crown damage. Sagebrush plants were cut with a fine-blade saw or sharp pruning shears at ground level (usually required removing accumulated detritus from around mature stems). Annual growth rings were field counted with a 10X or 20X hand lens (Ferguson 1964). It was often necessary to sample sagebrush plants outside the microplots to ensure that we had at least three estimates for each size class. Rotten stems or those damaged by grazing defied even approximate age determination in many plants. We frequently experienced stands where all sub-adult and adult specimens we sampled (10 to 20 specimens) could not be accurately aged; the smaller size classes had significant stem damage as well. The only burn information recorded was ignition source, either wildfire or prescribed burn, and the year of occurrence. We were unable to consistently characterize fire severity, a potentially significant explanatory variable because immediate post-fire conditions were not generally recorded at the time and quite a number of the burns were old enough (20+ years) that potentially telling clues had been obscured.

We estimated canopy cover (Daubenmire 1959) of all herbaceous species, bare soil, gravel, rocks, litter, lichens, mosses, and basal vegetation in the ten microplots using 13 cover classes ($T = >0, \leq 0.1$; $T = >0.1, \leq 1$; $P = >1, \leq 5$; $1 = >5, \leq 15$; $2 = >15, \leq 25$; $3 = >25, \leq 35$; $4 = >35, \leq 45$; $5 = >45, \leq 55$; $6 = >55, \leq 65$; $7 = >65, \leq 75$; $8 = >75, \leq 85$; $9 = >85,$

≤ 95 ; $F = >95, \leq 100$ percent). We also recorded the number of plants of each species of the Cichorieae tribe of the Asteraceae in each of the ten microplots. Taxonomic nomenclature follows the Plants Database (USDA Natural Resources Conservation Service 2008).

Data Analysis

We characterized Wyoming big sagebrush recovery as the ratio of mean canopy cover or height for the burned macroplot to the corresponding values from the unburned control macroplot. Sagebrush stand height was the mean of the size class with the greatest canopy cover. Rate of recovery for sagebrush was calculated as the percent recovery for either canopy cover or height divided by the number of years since burning. Species richness was measured by the number of vascular plant species recorded in the line intercepts and ten microplots.

The relative aridity of a site, as measured by precipitation and potential evapotranspiration, was hypothesized to affect recruitment and other aspects of stand recovery. Slope and aspect are the primary determinants of potential evapotranspiration; these two variables along with latitude have been integrated into a "heat loading" index by McCune and Keon (2002). Average site annual precipitation was estimated by DAYMET a statistical model that integrates elevation, other aspects of local terrain, and geographic position with weather station data for the past 20 years (Thornton and others 1997).

The Wilcoxon signed-rank test (statistic noted as *T*) was used to evaluate the differences between burned and unburned control macroplots for Wyoming big sagebrush canopy cover, total shrub cover and cover of perennial grasses, annual grasses, forbs and selected individual species; our data was congruent with the assumptions of this non-parametric test. Linear regression analysis was used to model the recovery of sagebrush height and canopy cover and herbaceous cover with time since fire; sagebrush recovery regression lines were forced through the origin to reflect biological realities. Regression analysis was also used to test the association between sagebrush recovery rate and the abiotic site factors of precipitation, heat load index and soil texture.

RESULTS

Wyoming Big Sagebrush and Shrub Recovery

The mean time since fire is 22 ± 16 ($16 = 1$ std. dev.) years, ranging from 4 to 67 years (median of 16 years) for the 24 sampled paired macroplots (control and burned). Fire resulted in virtually a complete loss of shrub canopy cover

as revealed by examination of recently burned macroplots (≤ 10 years, $N = 6$); five of the six plots had no shrub canopy cover and one had < 2 percent. Wyoming big sagebrush is the dominant shrub on the control macroplots with an average cover of 20 ± 8 percent, whereas on burned plots cover of this species averaged 0.81 ± 1.8 percent (difference highly significant, Wilcoxon's $T = 0$, $P = < 0.01$). Total shrub cover of control plots (21 ± 8 percent) averaged only slightly more than the cover of Wyoming big sagebrush alone, with the additional species most commonly including silver sage, rubber rabbitbrush (*Ericameria nauseosa*), green rabbitbrush (*Chrysothamnus viscidiflorus*) and spineless horsebrush (*Tetradymia canescens*). The difference in total shrub cover between control and burned plots (1.2 ± 2.1 percent) was highly significant ($N = 24$, Wilcoxon's $T = 0$, $P = < 0.01$). Silver sagebrush was the only shrub even approaching Wyoming big sagebrush in cover and that occurred on only one sandy loam site. The average height of the dominant cohort of Wyoming big sagebrush in control plots was 61 ± 11 cm.

A linear model resulted in the best fit for the recovery of Wyoming big sagebrush canopy cover with age since fire explaining 29 percent of the variation in cover ($t = 2.81$, $P = 0.010$; figure 2). Results for total shrub cover recovery were similar. The mean recovery rate for Wyoming big sagebrush canopy cover was 0.16 percent / year ± 0.45 for the first 70 years. Height recovery of the dominant Wyoming big sagebrush cohort was best fit by a highly significant ($t = 4.81$, $P = < 0.001$) exponential function (figure 3) in which time since fire explained 55 percent of the variation and the extrapolated time of recovery is approximately 68 years. We did not stratify the dataset according to mode of ignition because there were only six prescribed burn sites and four of these showed no recovery in big sagebrush cover; in addition fire effects, based on stump height, were the same between the two ignition modes. Of the 24 burned macroplots the seedling class was represented in only 4 macroplots, the juvenile class in 5 macroplots, the subadult class in 6 macroplots, and the adult class in 4 macroplots.

Heat load varied by only 12 percent between the most extreme of study sites, an expected result in rolling plains where the steepest slope study site slope was only 13 percent. Variation in understory herbaceous vegetation among the control plots was more likely due to the 50 percent difference in annual precipitation (10.8 to 16.3 inch [274 to 414 mm]) and soil texture. A linear regression model incorporating the heat load index and mean annual site precipitation explained 30 percent of the variation in the rate of Wyoming big sagebrush canopy recovery ($P = 0.064$). However, neither annual precipitation ($P = 0.827$) nor the heat index load ($P = 0.54$) alone were significantly related to canopy recovery rate.

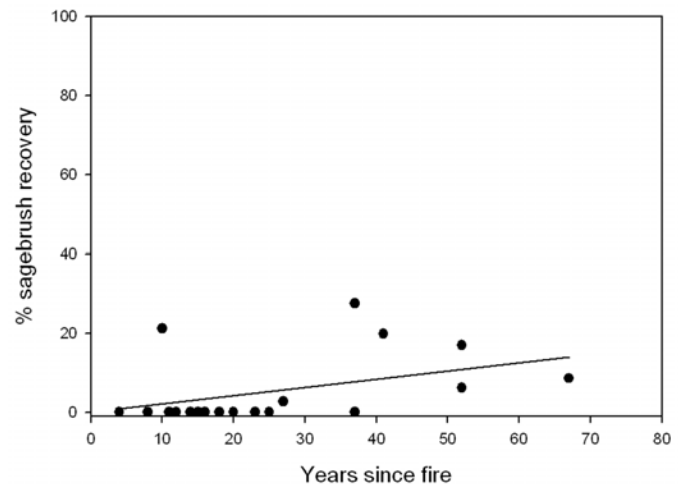


Figure 2—Linear regression model depicting Wyoming big sagebrush percent canopy recovery and time since fire for 24 sites (both wildfire and prescribed fire); regression model constrained to pass through the origin.

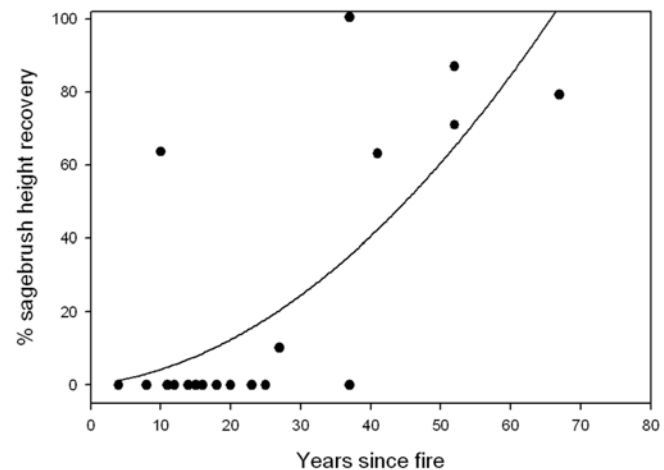


Figure 3—Exponential model depicting canopy height of Wyoming big sagebrush dominant cohort since fire for 24 sites; regression model constrained to pass through the origin.

Herbaceous Recovery

The important perennial graminoids in order of declining constancy were western wheatgrass, Sandberg bluegrass, blue grama, prairie junegrass (*Koeleria macrantha*), green needlegrass, needle-and-thread, bluebunch wheatgrass and sun sedge (*Carex inops* ssp. *heliophila*). Driest sites had relatively low canopy cover of bluebunch wheatgrass (mean = 28 percent cover), while undergrowth on the most mesic sites was dominated by Idaho fescue and western wheatgrass (mean = 74 percent cover). Grazing history probably also accounts for some of the variation in understory vegetation. Mean perennial grass canopy cover on control macroplots was 40 percent, of which approximately half was western wheatgrass. Burned macroplots had an average of 61 percent perennial grass cover, 39 percent of that cover is western wheatgrass. The differences in perennial grass cover between control and

burn macroplots was highly significant ($N = 24$, Wilcoxon's $T = 40$, $P = < 0.01$). Western wheatgrass had a mean of 39 percent and 22 percent cover in burned and control plots respectively ($N = 22$, Wilcoxon's $T = 19$, $P = < 0.01$). The other important vegetatively reproducing (sod-forming) graminoid, blue grama, exhibited 10 percent average canopy cover on burn plots and only 4 percent on controls, but due to highly variable cover this difference was not significant ($N = 21$, Wilcoxon's $T = 73$, $P = > 0.05$).

Annual grasses had an average canopy cover of 19 percent and 9 percent in burned and control macroplots respectively ($N = 24$, Wilcoxon's $T = 58$, $P = < 0.01$). Common species included the introduced brome grasses field brome (*Bromus arvensis*, formerly Japanese brome, *B. japonicus*) and cheatgrass (*Bromus tectorum*). However, the native sixweeks fescue (*Vulpia octoflora* = *Festuca octoflora*) also had appreciable constancy, although its cover was negligible.

There was no statistical difference in average canopy cover of forbs between burned (8.3 percent) and control (8.0 percent) macroplots ($N = 24$, Wilcoxon's $T = 89$, $P = > 0.05$). Though forb canopy cover was consistently less than graminoid cover, it did range as high as 27 percent due to an unusual post-fire increase in the non-native corn speedwell (*Veronica arvensis*) on one productive high-elevation site. The most common forbs were the non-natives pale madwort (*Alyssum alyssoides*), field cottonrose (*Logfia arvensis*, formerly *Filago arvensis*), herb Sophia (*Descurainia sophia*), littlepod false flax (*Camelina microcarpa*), yellow salsify (*Tragopogon dubius*), common dandelion (*Taraxacum officinale*), and the natives wooly plantain (*Plantago patagonica*), tiny trumpet (*Collomia linearis*), rough false pennyroyal (*Hedeoma hispida*), spiny phlox (*Phlox hoodii*), American vetch (*Vicia americana*) and scarlet globemallow (*Sphaeralcea coccinea*).

The most commonly occurring forbs of the *Cichorieae* tribe were the non-natives common dandelion and yellow salsify. The natives, prairie-dandelion (*Nothocalais troximoides*) and pale agoseris (*Agoseris glauca*), were found in only one and two plot pairs, respectively. Due to their extremely low densities native species were lumped with non-natives for analysis. The mean density of members of the *Cichorieae* tribe was 1.8 ± 2.8 plants / m^2 for the burned macroplots and 1.3 ± 1.8 plants / m^2 for the unburned controls; this difference was not significant ($N = 23$, Wilcoxon's $T = 132$, $P = >> 0.05$).

Species Richness

Species richness, the average number of species per macroplot, was 32 ± 5 for control plots and 26 ± 6 for burned plots and significantly different between treatments ($N = 24$, Wilcoxon's $T = 26$, $P = << 0.01$). Extreme values ranged from 19 to 44 and 12 to 40 for control and burned plots, respectively. An insignificant 0.7 percent ($R^2 =$

0.007) of the difference in species richness between burned and control macroplots was explained with time since burning ($t = 0.383$, $P = 0.71$).

DISCUSSION

Sagebrush and Shrubs

Observation of both recently burned stands and those of considerable post-burn age (> 20 years) indicate that Wyoming big sagebrush mortality was virtually complete. There was no measured canopy recovery for Wyoming big sagebrush in 17 of the 24 sites. Our linear model of canopy recovery based on 24 sample pairs indicates an exceedingly slow recovery rate. The highest sagebrush canopy recovery rate in our study, 0.72 percent / year still implies (assuming a linear model) full recovery would require about 138 years. An exponential function with recovery rate increasing with time is probably more realistic because it accounts for increasing propagule pressure Lesica and others (2007). Even so, the oldest burn, 67 years, was only 8 percent recovered and recovery on the most moisture-stressed sites as well as sites with the greatest precipitation and most mesophytic vegetation composition registered no recovery within 14 years. Even on an older (27 years), and ostensibly cooler prescribed burn, recovery was only 3 percent. The only site (Little Bighorn Battlefield National Monument) without domestic stock use within the recovery period (and for more than five decades prior to burning) recorded no shrub canopy recovery after 25 years.

Other studies have also found that Wyoming big sagebrush recovers slowly following fire. In the only other comparable study within our sampling area, Eichhorn and Watts (1984) found no re-establishment of Wyoming big sagebrush in 14 years following wildfires in the Missouri River Breaks and vicinity. In southwest Montana, Lesica and others (2007) documented almost no Wyoming big sagebrush canopy recovery in six wildfire burn plots, the most being 3 percent in 23 years. In southeastern Idaho (8.5 inch precipitation zone) Colket (2003) found, measuring density not cover, that 3 of 17 plots attained full recovery in 53 years and that by 92 years 16 of the 17 plots reached full density. As Baker (2007) observes attaining full density is not equivalent to recovery of canopy cover, which undoubtedly would require additional decades for shrubs to mature. However, some researchers have reported shorter recovery times. Wambolt and others (2001) reported a 72 percent recovery of Wyoming big sagebrush after 32 years in one burn and 96 percent recovery after only nine years in another in southwest Montana, and Watt and Wambolt (1996) documented 76 percent recovery within 30 years in another southwestern Montana study. It should be noted these studies documented recovery following cool-season, prescribed fires. A state-and-transition model (Hemstrom and others 2002) derived for the interior Columbia Basin

(ca. 8 inch annual precipitation) indicates that approximately 30 years are required to produce an open Wyoming big sagebrush / bunchgrass community starting with a post-burn bunchgrass-dominated habitat.

The average height of the dominant Wyoming big sagebrush cohort in control plots is 61 ± 11 cm, which agrees well with our southwestern Montana (Lesica and others 2007) measures of this subspecies (61 ± 6 cm). Only 4 of 24 burned macroplots even had a mature size/age class represented and the average height was 50 ± 6 cm; on one burned macroplot several individuals had attained full height recovery in approximately 38 years. Removing the zero values for height recovery from figure 3 would obviously shorten the time expected for full recovery and would more closely model what would be expected in the rate of height growth of individual plants once established on a site. However, for the model to be realistic on a stand basis the zero values should be included.

A nearby seed source is generally regarded as promoting faster stand recovery (Blaisdell 1953; Gruell 1980) because the seed bank of *A. tridentata* is negligible (Akinsoji 1988; Meyer 1992; Young and Evans 1989). Even though we located burn sample plots downwind from and as close as practicable to control plots, we still observed slow recovery rates. Wambolt and Payne (1986) found that close proximity to seed source still resulted in no Wyoming big sagebrush re-establishment six years post-burn.

We hypothesized that stands on areas of higher precipitation and/or with a lower heat load index would have a higher rate of recovery, similar to results from Johnson and Payne (1968). However, we were unable to detect any biotic or abiotic variables associated with Wyoming big sagebrush recovery across our study area. A model with age since fire, heat load index and precipitation explained 30 percent of the canopy recovery variation however, almost all of this explained variation was attributable to using age as a covariate.

More than 80 percent of burned macroplots lacked any representation of a seedling class. Seedling production was virtually nil, even in 2007, a year with abundant spring moisture that should have favored at least seedling germination, if not survival. We questioned this lack of seedlings as perhaps anomalous or a consequence of inadequate sampling. Therefore, in addition to visual scrutiny of the 10 microplots, we conducted extensive searches of adjacent terrain and uniformly failed to detect seedlings there as well. We encountered one instance of multiple seedling germination within a microplot (5 seedlings) out of 480 microplots examined. In general, the control macroplots had all maturity classes represented, however more than 50 percent of the stands did not have a seedling class present. The considerable difficulty Wyoming big sagebrush exhibits in site recolonization

might be expected given that it occupies the driest sites with the most poorly developed soils (Morris and others 1976; Barker and McKell 1983).

With the exception of the mostly missing seedling class, nearly all the unburned control plots were uneven-aged (had multiple size/maturity classes represented), indicating recruitment generally occurs in pulses. Three control plots were somewhat anomalous in that only an adult class was present. Two of these three plots had approximate stem ages indicating that there had been no recruitment in more than 25 and 40 years. Although the adult class of these two stands was not even-aged, their age structure suggests episodic reproduction at some point in time.

Graminoids

The large (21 percent) increase in perennial grass cover following fire showed no consistent diminution with time since fire, probably because the Wyoming big sagebrush canopy failed to make significant recovery even after more than 60 years. Possibly even greater increases in perennial grass cover may have been negated by post-fire livestock grazing when the grass becomes more accessible after shrub canopy elimination (Pechanec and others 1954; Harniss and Murray 1973; Bunting and others 1998). In our unburned control plots perennial grass cover, an index of long-term grazing intensity, was not associated with proportional changes in grass cover following fire. This implies that post-fire grazing has not had a large impact on fire-induced changes. The major contributor to the significant increase in post-fire perennial grass cover was the rhizomatous western wheatgrass. None of the other perennial grass species showed a consistent significant fire response although several species showed strong responses in one or two plots.

Average annual grass canopy cover for both burned (19 percent) and control (9 percent) macroplots ($N = 23$, $P = 0.010$) was comprised almost entirely of the non-native field brome, which exhibited no significant diminution of cover with time since fire. Its cover ranged from zero to 69 percent in burned plots. Field brome is usually regarded as a weed on rangelands because it competes with native perennials for water and nutrients (Stubbendieck and others 1985; Gartner and others 1976). Fire is noted (Gartner and others 1986; Whisenant 1990) to reduce field brome population density for one or two years post-burn primarily as a consequence of litter reduction (critical for seed germination and establishment). We found no research that followed this species post-burn course of succession for more than two years. We hypothesize that the observed field brome response was due to exploitation of space, water and nutrients following sagebrush mortality.

Forbs

Our results suggest that forbs are generally well-adapted to these fire-prone communities because there was no difference in cover between burn (8 percent) and control (8 percent) macroplots (paired-sample $t = .132$, $P = 0.896$). However, some plots did exhibit dramatic increases or decreases in forb cover, usually due to the unique responses of one or two species. For example, on both the youngest (4 years) and oldest (67 years) burns lesser spikemoss (*Selaginella densa*) had 0 to 2 percent cover in burn plots compared to 41 percent and 22 percent cover, respectively, on the control macroplots. On two plot pairs a positive response to burning was displayed by the annual non-natives field cottonrose (<1 percent to 24 percent) and corn speedwell (<1 percent to 23 percent). The rather stochastic nature of these responses is emphasized by the fact that field cottonrose cover was minor (1.5 percent) in the burned macroplot where corn speedwell cover was so high. It is noteworthy that these large differences in forb cover are due to annuals, not to native perennials, which register hardly any change. Similar results have been reported for prescribed burns in sagebrush steppe by Peek and others (1979), who found forb frequency was not affected three years post-burn, and also by the Harniss and Murray (1973) report of stable forb cover for 30 years following fire in eastern Idaho. Wildfire did not produce any change in canopy cover of forbs in south-central or southwest Montana (Hoffman 1996; Fraas and others 1992).

Forbs of the *Cichorieae* Tribe of the *Asteraceae* Family have been determined to comprise an important component of Greater Sage-Grouse summer diet and are often crucial for successful brood rearing (Klebenow and Gray 1967; Peterson 1970; Barnett and Crawford 1944; Drut and others 1994). An increase in forbs can be expected with the fire-induced reduction in the cover of shrubs and grasses (Klebenow 1973; Glenn-Lewin and others 1990). We combined the relatively rare occurrences (<40 plants / 1,440 m²) of native *Cichorieae*, prairie-dandelion and pale agoseris, with the much more abundant non-native *Cichorieae* densities, but found no evidence for a fire-driven change. Since non-native *Cichorieae* are invasive and increase with disturbance (Hobbs and Huenneke 1992; Kotanen and others 1998) their lack of response was unexpected. Lesica and others (2005) also found no change with fire for non-native *Cichorieae* in southwest Montana.

MANAGEMENT IMPLICATIONS

Most available evidence indicates that Wyoming big sagebrush canopy recovery rate is slow. Our data from central and southeastern Montana suggest that recovery (attaining the canopy cover exhibited by control plots) will

require much more than 100 years. No cases of “rapid” Wyoming big sagebrush recovery were observed within the study area. This observation is corroborated by limited data from southwestern Montana (Lesica and others 2007). The time to fully recover Wyoming big sagebrush on an extensive burn could be very considerable. Our results support the contention that, although fire is an important natural disturbance in sagebrush steppe, it could not have occurred as often as suggested in the past (see Baker 2007 for a review). Our results accord with Baker’s (2007) interpretation indicating that fire rotations are about 100 to 240 years for Wyoming big sagebrush and that sagebrush steppe belongs to fire regime V (long rotation, stand replacement). We found no evidence that soil texture, precipitation, slope, or were associated with the rate of canopy recovery in our study. Thus, managers cannot presume that stands of Wyoming big sagebrush on more mesic sites will necessarily exhibit faster recovery.

Our results are pertinent for managing domestic stock within the study area sagebrush steppe. We observed an average increase in perennial grass cover of 27 percent and 20 percent followed prescribed fire and wildfire, respectively. We have no evidence that this amplified cover will be diminished until sagebrush canopy cover becomes substantial at some future time, possibly at least a century after burning. Domestic livestock will benefit from this increased perennial grass cover and from increased accessibility to the herbaceous component due to shrub canopy removal. However, the 11 percent increase in annual grasses is due almost wholly to field brome which is considered by some a noxious weed (Stubbendieck and others 1985) because it competes with native perennials and has a brief window of grazing availability as it rapidly matures and loses nutrient content, digestibility and palatability (Stubbendieck and others 1985). Although various studies (see Stubbendieck and others 1985) indicate it declines with time on a site we have no indication this is the case. Burning sagebrush stands infested with field brome may result in a long-term increase in this undesirable species.

These results also have implications for protecting sagebrush-dependent wildlife. Land managers should be aware of Wyoming big sagebrush post-fire response. Greater Sage-grouse are dependent on some mixture of open- and closed-canopy sagebrush habitats to complete their life cycle (Connelly and others 2000). Wyoming big sagebrush recovery takes so long that managers considering prescriptive burns need to have a long-term view of the landscape before eliminating a sagebrush habitat that will probably not be sufficiently dense to constitute appropriate habitat for at least 50 years and perhaps much longer. Even greater concern should attend wildfire management in sagebrush habitats, if for no other reason than the resulting burns usually are more extensive (than prescribed burns) and hence result in yet greater recovery times. Greater

Sage-Grouse should be benefited by increased perennial grass cover resulting from fire (Wallestad and Pyrah 1974; Aldridge and Brigham 2002). Although success of Greater Sage-grouse brood rearing is dependent on available forbs, especially those of the Cichorieae, both native and exotic (Connelly and others 2000), we found no predictable increase in forb cover, including those of the Cichorieae, with fire.

Meyer's (1992) thorough account of the ecology of sagebrush recruitment notes aerial seeding (fall or winter), should be considered a potentially cost-effective approach for sagebrush establishment; however, this approach is subject to stochastic effects (and failure) beyond the manager's control as documented by Blew and others (2004). On less extensive burns nucleating with snow harvesting sites (creating favorable microsites) shows potential for enhancing emergence and recruitment (Meyer 1992).

CONCLUSION

Wyoming big sagebrush recovery from both prescribed fire and wildfire was extremely slow within the study area, likely requiring well over 100 years to reach pre-burn sagebrush cover. Results were similar across all environmental conditions, even on relatively mesic sites. Perennial and annual grass cover increased after burning, but the annual grass increase consisted almost entirely of field (Japanese) brome, a non-native that is considered a weed with negative habitat and livestock value. Forbs, most especially those of the Cichorieae tribe of the Asteraceae family, are important for Greater Sage-grouse brood rearing; however, we found no predictable change of this component with fire. Plant species richness was lower in burned plots. Resource managers concerned about Greater Sage-grouse and other sage-dependent species should carefully consider the long-term ramifications of any form of burning on Wyoming big sagebrush habitat in eastern Montana.

ACKNOWLEDGEMENTS

The following were instrumental in locating sampling sites, Louise deMontigny and Eric Lepisto (Bureau of Land Management, Miles City Field Office), Jeff Dibendetto (ecologist, Custer National Forest, Billings), Michael Stops (Chief Ranger, Little Bighorn Battlefield National Monument), Tom Stivers (wildlife biologist, Montana Fish Wildlife and Parks, Lewistown) and Bill Berg, Bob Skinner and Joann Dullum (Charles M. Russell National Wildlife Refuge, Lewistown). We extend our sincere appreciation to field assistants Lisa Wilson and David Salazar. Larry Eichhorn (retired BLM range conservationist, Lewistown) was generous with his time, helping us relocate a number of his original sagebrush succession study sites and dispensing

informed commentary on our study. Soils analyses were performed at the facilities of Oregon State University in Corvallis, OR. We thank the Bureau of Land Management for financial support of this project.

REFERENCES

- Akinsoji, A. 1988. Postfire vegetation dynamics in a sagebrush steppe in southeastern Idaho, USA. *Vegetatio* 78: 151-155.
- Aldridge, C.L., Brigham, R.M. 2002. Sage grouse nesting and brood habitat use in southern Canada. *Journal of Wildlife Management*. 66: 433-444.
- Antos, J.A., McCune, B., Bara, C. 1983. The effect of fire on an ungrazed western Montana grassland. *American Midland Naturalist*. 110: 354-364.
- Bailey, R.G. 1995. Descriptions of the ecoregions of the United States. Second Edition Revised and Expanded. USDA, Forest Service. Misc. Publ. 1391. 108 p.
- Baker, W.L. 2007. Fire and restoration of sagebrush ecosystems. *Wildlife Society Bulletin*. 34(1): 177-185.
- Barker, J.R., McKell, C.M. 1983. Habitat differences between basin and Wyoming big sagebrush in contiguous populations. *Journal of Range Management*. 36: 450-454.
- Barnett, J.K., Crawford, J.A. 1994. Pre-laying nutrition of sage grouse hens in Oregon. *Journal of Range Management*. 47:114-118.
- Blaisdell, J.P. 1953. Ecological effects of planned burning of sagebrush-grass range on the upper Snake River Plains. Washington, DC. USDA Technical Bulletin 1075. 39 p.
- Blew, R., Forman, A., Hafla, J., Pellant, M., Jones, R., White, G., Sands, A., Klar, T. 2004. Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) recovery following fire and artificial seeding. 2 p. Available: <http://abstracts.co.allenpress.com/pweb/esa2004/document/?ID=37567> [December 28, 2007].
- Bunting S.C., Robberecht, R., Defossé, G.E. 1998. Length and timing of grazing on postburn productivity of two bunchgrasses in an Idaho experimental range. *International Journal of Wildland Fire*. 8: 15-20.
- Clifton, N.A. 1981. Response to prescribed fire in a Wyoming big sagebrush / bluebunch wheatgrass habitat type. Moscow: University of Idaho; 39 p. MS Thesis.
- Colket, E.C. 2003. Long-term vegetation dynamics and post-fire establishment patterns of sagebrush steppe. Moscow: University of Idaho; 144 p. MS Thesis.
- Connelly, J.W., Schroeder, M.A., Sands, A.R.; Braun, C.E. 2000. Guidelines to manage sage grouse populations and habitat. *Wildlife Society Bulletin*. 28: 967-985.
- Daubenmire, R. 1959. A canopy-coverage method of vegetational analysis. *Northwest Science*. 33: 43-64.
- Drut, M.S., Pyle, W.H., Crawford, J.A. 1994. Diets and food selection of sage grouse chicks in Oregon. *Journal of Range Management*. 47: 90-93.
- Eichhorn, L.C., Watts, C.R. 1984. Plant succession on burns in the river breaks of central Montana. *Proceedings of the Montana Academy of Science*. 43: 21-34.
- Eng, R.L., Schladoweiller, P. 1972. Sage grouse winter movements and habitat use in central Montana. *Journal of Wildlife Management*. 36: 141-146.
- Ferguson, C.W. 1964. Annual rings in big sagebrush. *Papers of the Laboratory of Tree-Ring Research* No. 1. Tucson: University of Arizona Press. 95 p.
- Fraas, W.W., Wambolt, C.L., Frisina, M.R. 1992. Prescribed fire effects on a bitterbrush-mountain big sagebrush-bluebunch wheatgrass community. *Proceedings of a symposium*; 1992: Ogden, UT. Gen. Tech. Rep. INT-289. Ogden, UT: USDA, Forest Service, Intermountain Forest Range Experiment Station. 232 p.
- Gartner, F.R., White, E.M., Butterfield, R.I. 1986. Mechanical treatment and burning for high quality range forage. *Beef Report: Cattle* 86-29. Brookings, SD: South Dakota State University, Department of Animal and Range Sciences and Plant Science; Agriculture Experiment Station. 140 p.
- Gartner, F.R., Roath, L.R., White, E.M. 1976. Advantages and disadvantages of prescribed burning. *Proceedings of a symposium*; 1976; Logan, UT. Use of prescribed burning in western woodland and range ecosystems: Utah State University: 11-15 p.

- Glenn-Lewin, D.C., Johnson, L.A., Jurik, T.W., Kosek, A., Leoscheke, M., Rosburg, T. 1990. Fire in central North American grasslands: vegetative reproduction, seed germination and seedling establishment. In S.L. Collins and L.L. Wallace editors, Fire in central North American grasslands. Norman: University of Oklahoma Press. 28-45 p.
- Gruell, G.E. 1980. Fire's influence on wildlife habitat on the Bridger-Teton National Forest, Wyoming, Volume II- changes and causes, management implications. Ogden, UT: Res. Pap. INT-252. USDA, Forest Service, Intermountain Forest and Range Experiment Station. 35 p.
- Harniss, R.O., Murray, R.B. 1973. 30 years of vegetal change following burning of sagebrush-grass range. Journal of Range Management. 26: 322-325.
- Hemstrom, M.A., Wisdom, M.J., Hann, W.J., Rowland, M.M., Wales, B.C., Gravenmier, R.A. 2002. Sagebrush-steppe vegetation dynamics and restoration potential in the Interior Columbia Basin. Conservation Biology. 16: 1243-1255.
- Hobbs, R.J., Huenneke, L.F. 1992. Disturbance, diversity and invasion: implications for conservation. Conservation Biology. 6: 324-337.
- Hoffman, T.L. 1996. An ecological investigation of mountain big sagebrush in the Gardiner Basin. Bozeman: Montana State University; 84 p. M S Thesis.
- Humphrey, L.D. 1984. Patterns and mechanisms of plant succession after fire on *Artemisia*-grass sites in southeastern Idaho. Vegetatio. 57: 91-101.
- Johnson, J.R., Payne, G.F. 1968. Sagebrush reinvasion as affected by some environmental influences. Journal of Range Management. 21: 209-213.
- Klebenow, D.A. 1969. Sage grouse nesting and brood habitat in Idaho. Journal of Wildlife Management. 33: 649-661.
- Klebenow, D.A. 1973. The habitat requirements of sage grouse and the role of fire in management. Proceedings of the Tall Timbers Fire Ecology Conference. 12: 305-315.
- Klebenow, D.A., Gray, G.M. 1967. Food habits of juvenile sage grouse. Journal of Range Management. 21: 80-83.
- Kotanen, P.M., Bergelson, J., Hazlett, D.L. 1998. Habitats of native and exotic plants in Colorado shortgrass steppe: a comparative approach. Canadian Journal of Botany. 76: 664-672.
- Lesica, P., Cooper, S.V., Kudray, G. 2005. Big sagebrush shrub-steppe postfire succession in southwest Montana. Helena, MT; Montana Natural Heritage Program. Report to USDI, Bureau of Land Management, Dillon Field Office. 29 p. plus appendices.
- Lesica, P., Cooper, S.V., Kudray, G. 2007. Recovery of big sagebrush following fire in southwest Montana. Rangeland Ecology & Management. 60: 261-269.
- McCune, B., Keon, D. 2002. Equations for potential annual direct incident radiation and heat load. Journal of Vegetation Science. 13: 603-606.
- Meyer, S.E. 1992. Germination and establishment ecology of big sagebrush: Implications for community restoration. Proceedings of a symposium; 1992 May18-21; Boise, ID. Gen. Tech. Rep. INT-GTR-313. Ogden, UT: Forest Service, Intermountain Research Station. 416 p.
- Morris, M.S., Kelsey, R.G., Griggs, D. 1976. The geographic and ecological distribution of big sagebrush and other woody *Artemisias* in Montana. Proceedings of the Montana Academy of Sciences. 36: 56-79.
- Mueller-Dombois, D., Ellenberg, H. 1974. Aims and methods of vegetation ecology. New York: John Wiley & Sons. 547 p.
- Nesser, J.A., Ford, G.L., Maynard, C.L., Page-Dumroese, D.S. 1997. Ecological units of the Northern Region: subsections. General Technical Report INT-GTR-369. Ogden, UT: USDA, Forest Service, Intermountain Research Station. 88 p.
- Pechanec, J., Stewart, F.G., Blaisdell, J.P. 1944. Sagebrush burning-good and bad. USDA Farmer's Bulletin 148. 34 p.
- Peck, J.M., Riggs, R.A., Lauer, J.L. 1979. Evaluation of fall burning on bighorn sheep winter range. Journal of Range Management. 32: 430-432.
- Peterson, J.G. 1970. The food habits and summer distribution of juvenile sage grouse in central Montana. Journal of Wildlife Management. 34: 147-155.
- Pyle, W.H., Crawford, J.A. 1996. Availability of foods for sage grouse chicks following prescribed fire in sagebrush-bitterbrush. Journal of Range Management. 49: 320-324.
- Roscoe, J.W. 2002. Sage grouse movements in southwestern Montana. Intermountain Journal of Science. 8: 94-104.
- Stubbendieck, J., Nichols, J.T., Roberts, K.K. 1985. Nebraska range and pasture grasses (including grass-like plants). E.C. 85-170. Lincoln, NE: University of Nebraska, Department of Agriculture, Cooperative Extension Service. 75 p.
- Thornton, P.E., Running, S.W., White, M.A. 1997. Generating surfaces of daily meteorological variables over large regions of complex terrain. Journal of Hydrology. 190: 214-251.
- USDA Natural Resource Conservation Service. 2008. Plants Database [http://plants.usda.gov/index.html.]
- Walhof, K.S. 1997. A comparison of burned and unburned big sagebrush communities in southwest Montana. Bozeman: Montana State University; 74 p. MS Thesis.
- Wallestad, R. 1971. Summer movements and habitat use by sage grouse broods in central Montana. Journal of Wildlife Management. 35:129-135.
- Wallestad, R., Pyrah D. 1974. Movement and nesting of sage grouse hens in central Montana. Journal of Wildlife Management. 38: 630-637.
- Wambolt, C.L., Payne, G.F. 1986. An 18-year comparison of control methods for Wyoming big sagebrush in southwestern Montana. Journal of Range Management. 39: 314-319.
- Wambolt, C.L., Walhof, K.S., M.R. Frisina. 2001. Recovery of big sagebrush communities after burning in south-western Montana. Journal of Environmental Management. 61: 243-252.
- Watts, M.J., Wambolt, C.L. 1996. Long-term recovery of Wyoming big sagebrush after four treatments. Journal of Environmental Management. 46: 95-102.
- Welch, B.L., Cridle, C. 2003. Countering misinformation concerning big sagebrush. Research Paper RMRS-RP-40. Ogden, UT: USDA, Forest Service, Rocky Mountain Research Station. 28 p.
- West, N.E. 2000. Synecology and disturbance regimes of sagebrush steppe ecosystems. Pages 15-26 in Entwistle, P.G., DeBolt, A.M., Kaltenecker, J.H., Steenhof, K. compilers. Proceedings of a symposium; 1999 June23-25; Boise, ID. Publication No. BLM/ID/PT-001001+1150. 145 p.
- West, N.E., Hassan, M.A. 1985. Recovery of sagebrush-grass vegetation following wildfire. Journal of Range Management. 38: 131-134.
- Whisenant, S.G. 1990. Postfire population dynamics of *Bromus japonicus*. American Midland Naturalist. 123: 301-308.
- Winward, A.H. 1991. A renewed commitment to management of sagebrush grasslands. In Research in rangeland management. Corvallis, OR: Oregon State University Agricultural Experiment Station Special Report 880. 7 p.
- Winward, A.H. 2004. Sagebrush of Colorado: Taxonomy, distribution, ecology and management. Denver, CO: Colorado Division of Wildlife, Department of Natural Resources. 45 p.
- Wright, H.A., Neuenschwander, L.F., Britton, C.M. 1979. The role and use of fire in sagebrush-grass and pinyon-juniper plant communities: A state-of-the-art review. Gen. Tech. Rep. INT-58. Ogden, UT: USDA, Forest Service, Intermountain Forest and Range Experiment Station. 48 p.
- Young, J.A., Evans, R.A. 1989. Dispersal and germination of big sagebrush (*Artemisia tridentata*) seeds. Weed Science. 37: 201-206.

Dynamics of Western Juniper Woodland Expansion into Sagebrush Communities in Central Oregon

Mary M. Rowland¹, Lowell H. Suring², Robin J. Tausch³,
Susan Geer⁴, and Michael J. Wisdom⁵

ABSTRACT

Western juniper (*Juniperus occidentalis*) woodlands in Oregon have expanded four-fold from 600,000 ha in 1930 to > 2.6 million ha, often resulting in the reduction and fragmentation of sagebrush (*Artemisia* spp.) communities. We documented dynamics of western juniper across the John Day Ecological Province in central Oregon by recording size class and growth form at 178 sites. We used stratified random sampling, with strata based on vegetation association (sagebrush, juniper, other) and distance from juniper stands. Only 26 percent of sites contained pre-settlement trees (in other words, > 140 years old), and < 5 percent of the 2,254 junipers tallied were pre-settlement trees. Mean densities of pre-settlement trees by stratum ranged from 0 to 18 trees/ha, suggesting that historically, juniper was widely scattered across the landscape. Current densities of post-settlement trees ranged from 75 to 211 trees/ha in non-woodland strata to 457 trees/ha in the juniper stratum. Juniper in non-woodland strata was most abundant in sites adjacent to juniper stands and in sagebrush communities. Mean densities of post-settlement trees were greatest in the > 2.0-m tall size class (82 trees/ha), followed by the 0.3 to 1-m tall size class (52 trees/ha). These densities pose substantial risk to sagebrush communities in central Oregon. Questions remain about the extent of western juniper woodlands across the species' range that have replaced or are expanding into sagebrush communities versus sites that historically supported woodlands. However, our findings suggest that within sagebrush communities of the John Day province, intensive management through removal of western juniper may be prudent, while retaining pre-settlement trees.

INTRODUCTION

A striking change in landscapes of the western United States during the past 100 to 150 years has been the expansion of pinyon-juniper (*Pinus* L. – *Juniperus* L.) woodlands into the sagebrush (*Artemisia* spp.) ecosystem (Chambers 2001; Miller and Rose 1995; Miller and

Tausch 2001; Miller and others 1999a; figure 1) and, to a lesser extent, grassland ecosystems (Coppedge and others 2004; Miller and others 2005; Taylor 2007). Pinyon-juniper woodlands have increased 10-fold in extent throughout the Intermountain West since the late 1800s and currently encompass at least 30 million ha (Miller and Tausch 2001). These woodlands are likely to continue to expand because conditions under which these communities thrive include currently unoccupied areas (Betancourt 1987; West and Van Pelt 1987). Pinyon and juniper species can be successional aggressive across their range and, once they invade, can eliminate the understory component of a community (Azuma and others 2005; Johnsen 1962; Miller and others 2000; Tausch and Tueller 1990).



Figure 1—Encroaching western juniper in a big sagebrush community near Mitchell, Oregon.

Existing pinyon-juniper woodlands have been shaped by long-term interactions between environmental conditions, including climate and current vegetation (Miller and Wigand 1994; Tausch 1999). Several events during the last 150 years led to increased rates of woodland establishment throughout the Intermountain West. These include: climate change, with rising temperatures; intensive livestock use; decreased wildfire and increased fire suppression; and increasing atmospheric CO₂ that benefited the dominance of large, woody perennial plants (Brown and Smith 2000; Clark and others 1998; Gedney and others 1999; Miller and others 2005; 2008; Miller and Rose 1995; Soule and others 2003; Tausch 1999). The risk of crown fire in these increasingly dense woodlands is perhaps greater than at any time since the Neoglacial period (Miller and others 2008; Tausch 1999).

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Mary M. Rowland, Wildlife Biologist, Pacific Northwest Research Station, La Grande, OR, mrowland@fs.fed.us

²Lowell H. Suring, Wildlife Ecologist, Forest Service, Terrestrial Wildlife Ecology Unit, Boise, ID

³Robin J. Tausch, Research Range Scientist, USDA, Forest Service, Rocky Mountain Research Station, Reno, NV

⁴Susan Geer, Botanist, USDA Forest Service, Pacific Northwest Research Station, La Grande, OR

⁵Michael J. Wisdom, Research Wildlife Biologist, USDA Forest Service, Pacific Northwest Research Station, La Grande, OR

Once established, woodlands may provide wood products (Pieper and others 2002) and habitat for many wildlife species (for example, Belsky 1996; Dobkin and Sauder 2004; Miller and others 2005; Pavlacky and Anderson 2001; Rumble and Gobeille 1995), and potentially serve as carbon sinks (Hubbard and others 2003). However, conversion of sagebrush and other native plant communities to pinyon-juniper woodlands places additional stress on communities that have been severely reduced in area and habitat quality (Connelly and others 2004; Miller and others 1999a, Schaefer and others 2003). These landcover conversions may have profound ecological implications (Huxman and others 2005).

Concurrent with and partially due to woodland expansion, the sagebrush ecosystem in western North America has dramatically declined in quality and quantity (Bunting and others 2002; Knick 1999; Miller and Eddleman 2000; Wisdom and others 2005). Sagebrush-associated fauna, such as greater sage-grouse (*Centrocercus urophasianus*), pygmy rabbit (*Brachylagus idahoensis*), and Brewer's sparrow (*Spizella breweri*), are considered species of concern across much of their range (Connelly and others 2004; Dobkin and Sauder 2004; Knick and others 2003; Rowland and others 2005). Moreover, the threat of woodland encroachment was determined to be an extinction risk for greater sage-grouse in the western portion of its range (USDI Fish and Wildlife Service 2005). Connelly and others (2004) also identified pinyon-juniper woodland encroachment as a threat in their range-wide conservation assessment of greater sage-grouse habitats. However, old-growth pinyon-juniper, which should be distinguished from expansion woodlands, provides essential habitat for many woodland-associated species of conservation concern (Miller and others 1999b, Reinkensmeyer and others 2008).

In addition to displacing plant communities such as sagebrush and being implicated in the increasing distribution of invasive plants such as cheatgrass (*Bromus tectorum*),¹ encroaching woodlands also increase fuel loads, thereby leading to changes in fire regimes (Chambers and others 2005). Pinyon and juniper species are highly flammable and vulnerable to fire (Brown and Smith 2000). The issue of pinyon-juniper expansion is particularly problematic because the phenomenon is geographically widespread and is a potentially divisive issue among public land users (for example, Nelson and others 1999). Woodland expansion is also considered a threat to effective management of forage resources for livestock (Bates and others 2000; Gholz 1980). In Oregon, pinyon-juniper woodlands comprise a single species, western juniper (*Juniperus occidentalis*) (hereafter referred to as juniper). This species represents the northwestern extension of pinyon-juniper woodlands in the Intermountain West (Miller and others 2005). Juniper woodlands have expanded four-fold in Oregon since 1930, from 600,000 ha to more than 2.6 million ha (Azuma and others 2005). Juniper

grows on xeric lands in Oregon, generally at elevations between 900 and 1,500 m and in locations with 25 to 50 cm annual precipitation (Azuma and others 2005; Gedney and others 1999). In Oregon, juniper is typically classified as either juniper forest (> 10 percent crown cover of juniper) or savanna (juniper present but < 10 percent crown cover and < 5 percent stocking) (Azuma and others 2005; Gedney and others 1999). As of 1999, Oregon had an estimated 1.35 million ha of juniper forest and 1.31 million ha of juniper savanna. The extent of juniper forest in Oregon is predicted to increase as small junipers grow and juniper savannas convert to forests (Azuma and others 2005).

Our primary objective in this paper is to describe the dynamics of juniper establishment in existing sagebrush communities of central Oregon, specifically by describing densities, age, and height classes of juniper measured in sagebrush-dominated sites. A second objective is to describe the structure of existing juniper stands in central Oregon. The results described below are part of a comprehensive, ongoing project to evaluate a model to estimate risk of woodland encroachment into sagebrush with field and remotely sensed data (Suring and others 2005). By understanding the current and predicted extent of juniper encroachment into sagebrush, managers can assume a balanced approach to woodland management that maintains old-growth juniper while engaging in active restoration of sagebrush communities where prudent and feasible.

METHODS

Study Site

Eastern Oregon

The climate of eastern Oregon, from the crest of the Cascade Mountains eastward, is highly variable because of maritime, continental, and arctic influences (Ferguson 2001). Temporal and spatial variability in climate is evident at watershed scales throughout eastern Oregon due to complex topography and mosaics of land cover (Mock 1996). Median annual minimum temperatures for eastern Oregon vary from -18° C in the more protected areas of the Columbia Basin to -32° C in the high mountains and plateaus. July mean temperatures range between 18° C and 21° C in the central valleys and plateaus and up to 26° C along the eastern border with Idaho. Precipitation is generally low, ranging from 10 to 50 cm annually (Campbell and others 2003).

Thirty-five percent (> 6 million ha) of eastern Oregon is forested (in other words, ≥ 10 percent forest cover); the remainder supports grasses and shrubs (Campbell and others 2003). Several tree species are abundant at higher elevations (> 1200 m) including ponderosa pine (*Pinus ponderosa*), lodgepole pine (*Pinus contorta*), Douglas-fir (*Pseudotsuga menziesii*), white fir (*Abies concolor*), grand

fir (*Abies grandis*), and quaking aspen (*Populus tremuloides*) (Franklin and Dyrness 1973). At lower elevations (760 to 1400 m), western juniper savannas occur in a zone between ponderosa pine forests and shrub and grass communities. Within the shrub communities dominant species include big sagebrush (*Artemisia tridentata*), antelope bitterbrush (*Purshia tridentata*), scabland sagebrush (*A. rigida*), low sagebrush¹ (*A. arbuscula*), and shadscale saltbush (*Atriplex confertifolia*). Within the grassland communities dominant species include bluebunch wheatgrass (*Pseudoroegneria spicata*), Idaho fescue (*Festuca idahoensis*), basin wildrye (*Elymus cinereus*), Thurber's needlegrass (*Stipa thurberiana*), and the invasive annual grass cheatgrass.

John Day Ecological Province²

We used the ecological provinces from Miller and others (1999a) and West and others (1998) as the geographic basis for development of rules for the woodland model and associated data collection and analysis. Ecological provinces are large areas (in other words, millions of ha), each of which is defined by similarity in regional climate, topography, potential natural communities, geology, and soils (Anderson and others 1998; Suring and others 2005; West and others 1998). The ecological characterization of landscape conditions within each of the provinces provided a useful and important ecological context for describing juniper relationships with environmental factors (for example, Miller and others 2008).

We selected the John Day province for our analysis because of its diversity of sagebrush communities and the widespread distribution of western juniper (figure 2). The John Day province extends across much of east-central Oregon, encompassing 3.5 million ha with diverse land covers ranging from coniferous forests to expansive shrub and grass communities. The province is topographically diverse with valleys and dissected hills interspersed with taller buttes and plateaus. Watersheds of the Crooked River and John Day River (with the exception of the Upper North Fork) are contained within the John Day province. Elevation ranges from 335 to 3,060 m ($\bar{x}$ = 1,262 m), with the highest elevations in the Ochoco Mountains. Average annual precipitation across the province is 48 cm. Over half of the rainfall occurs between November and March. Soils in the province are primarily fine-textured and calcareous.

¹Plant nomenclature follows USDA Natural Resources Conservation Service (2008). One exception is our use of the common name “low sagebrush” for *Artemisia arbuscula*, instead of the recommended name “little sagebrush.”

²See Anderson and others (1998) for a more in-depth description of the John Day province and for descriptions of other ecological provinces in Oregon.

The LANDFIRE existing vegetation type (EVT) map is one of a suite of nationwide grids developed to provide consistent mapping of ecosystems, fuels, and fire across the U.S., with a 30-m (0.09 ha) resolution (Rollins and Frame 2006). We selected LANDFIRE to describe current land cover for our analysis due to its national coverage and consistency. Ninety-two “ecological systems” (a mid-scale vegetation classification system, between fine-grained ecological communities and coarse-grained ecoregions; see Comer and others [2003]) occur in the province, as mapped by LANDFIRE. Most (76), however, each comprise < 1 percent of the study area. Coniferous forests dominate the vegetation, with ponderosa pine and mixed dry conifer forests together covering nearly 30 percent of the province.

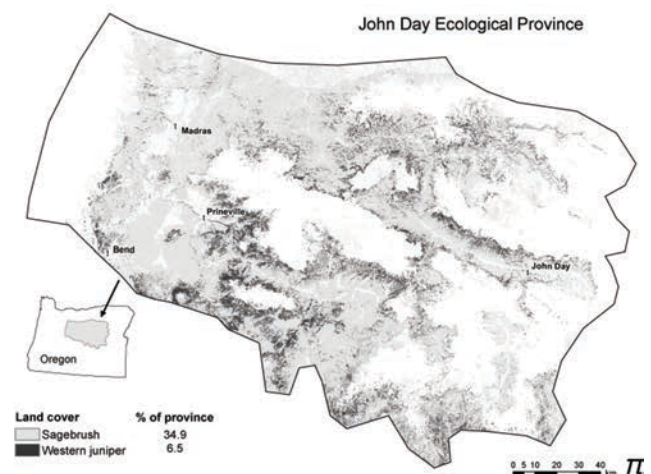


Figure 2—Western juniper and sagebrush (all sagebrush ecological systems combined) as mapped by LANDFIRE in the John Day Ecological Province, central Oregon. See text for details.

The most abundant (550,000 ha; 15.6 percent) single vegetation type is Inter-mountain Basins Big Sagebrush Steppe, which is a widespread, matrix-forming system (in other words, a patch type forming extensive and continuous cover; [Comer and others 2003]) composed primarily of Wyoming big sagebrush (*Artemisia tridentata wyomingensis*) and basin big sagebrush (*A. t. tridentata*). Other common sagebrush types in the John Day province include mountain big sagebrush (*A. t. vaseyana*), low sagebrush, and scabland sagebrush. All sagebrush types combined cover 1.23 million ha (34.9 percent) of the province (figure 2). The majority of sagebrush is on private lands (60.7 percent), which is somewhat higher than expected given the extent of private lands in the province (47.2 percent; figure 3). The same is true for the percentage of sagebrush managed by the USDI Bureau of Land Management (BLM) (21.2 percent) compared to the extent of BLM-administered lands in the province (13.0 percent). The USDA Forest Service manages only 8 percent of the sagebrush (16,180 ha; much of this is mountain big sagebrush) but manages 32.8 percent of the John Day province.

The area is mostly rural, with human populations centered at Bend, John Day, Madras, and Prineville. Livestock production, agriculture, extraction of minerals, and timber harvest on private and public lands provide the basis for local economies and, as a result, strongly influence the social structure of the study area.

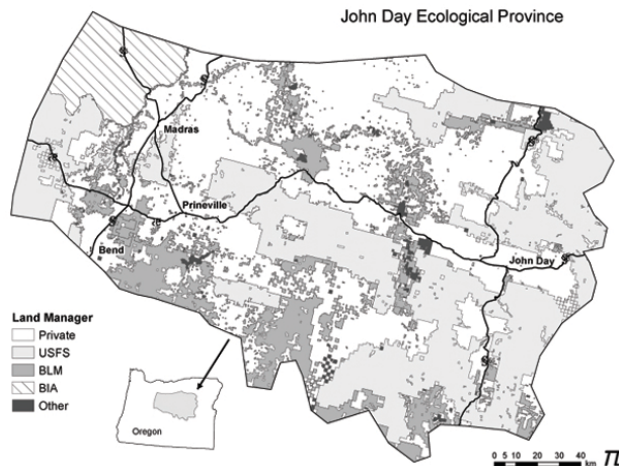


Figure 3—Location of the John Day Ecological Province in central Oregon. Land managers are as follows: USFS = USDA Forest Service, BLM = USDI Bureau of Land Management, BIA = USDI Bureau of Indian Affairs, and Other, including Oregon Department of State Lands and USDI Fish and Wildlife Service.

Sampling Design

Sample Stratification

The target frame for field sampling was any pixel of a sagebrush land cover type (as delineated by the 30-m pixels in the LANDFIRE EVT map; figure 2) within the John Day province. The sampling frame was considerably smaller, however, due to logistical constraints associated with collection of field data (for example, plots were located < 1 km of a road to decrease travel time to plots). To ensure that a representative sample was obtained from a range of environmental conditions, we used a stratified random approach. Due to the limited number of plots that we estimated could be sampled during the field season (~200), we formed six sampling strata representing combinations of two key variables—land cover type (3 classes) and proximity to juniper stands³ (2 classes) (table 1). We added a 7th stratum, juniper stands, to obtain estimates of tree density and size distribution in sites mapped as existing western juniper by LANDFIRE, for comparison with juniper conditions in sagebrush sites at various stages of juniper encroachment.

Non-woodland (in other words, all but juniper) ecological systems were collapsed into three types for the stratification: mountain big sagebrush/low sagebrush, other sagebrush, and non-sagebrush (table 1). We stratified by

³A “stand” of juniper was defined as ≥ 10 ha of contiguous juniper pixels for this analysis.

mountain big sagebrush, despite its rarity (1.8 percent, estimated using LANDFIRE) in the province, because it is a key sagebrush type, especially as brood-rearing habitat for greater sage-grouse, and is typically the dominant sagebrush taxon on National Forest System lands. Also, a review of BLM sagebrush plot data (~1200 records for the province) within sites mapped as western juniper revealed that most of the records reported mountain big sagebrush as the dominant shrub. Thus, this type may be important in the sagebrush/juniper ecotone in our study area.

We used two buffer zones extending out from juniper stands (1 – 1600 m and 1600 – 5000 m) to create strata delineating proximity to juniper stands and thereby describe dynamics of juniper encroachment (table 1). Juniper seeds are commonly dispersed ≤ 1.6 km from juniper stands by birds and mammals (Schupp and others 1999). Birds have also been reported to disperse seeds up to 5 km from seed sources, but in lesser amounts (Vander Wall and Balda 1977, 1981).

We delineated these seven strata across the John Day province with a geographic information system (GIS) (table 1). We then applied a series of filters to remove non-targeted or inaccessible areas from the sampling frame as follows:

1. Private or tribal lands (to facilitate access);
2. Areas < 100 m of an accessible (in other words, non-private) road (to ensure that no portion of the sample plot fell within ~50 m of a road and associated disturbance effects);
3. Areas > 1.0 km from accessible roads (to reduce travel time to plots);
4. The Crooked River National Grassland (because of extensive, long-term land management actions, such as juniper removal and seeding projects); and
5. Areas that were not homogeneous in terms of stratum type; if < 6 pixels out of 9 in a 3 x 3 neighborhood were classified the same as the center pixel, the pixel was removed from the sampling pool.

Within the area remaining after the above filters were applied, 75 random points were generated in each of the seven strata using a GIS. If a random point was assigned to a pixel within 200 m of a previously generated point, the point was dropped and a new point generated in the GIS. This step eliminated sampling of points that were too close together to represent independent samples of woodland condition, which we characterized at a scale of ~1 ha with our sampling design. The resulting spatial distribution of sample points was examined to ensure that an adequate geographic representation of sagebrush across the province was captured.

The targeted number of plots to sample within each of the strata was determined by a weighting factor that incorporated both the proportion of available pixels in the stratum and the estimated variance of juniper densities within that stratum (table 1). For example, 71 of the estimated 200 plots to be sampled were allocated to the stratum containing “other sagebrush, far from juniper stands,” due to the complex of sagebrush taxa that could occur in the “other sagebrush” type as well as the higher uncertainty about juniper densities with increasing distance from juniper stands. We also pre-screened plots in the two non-sagebrush strata using digital aerial photography to eliminate sampling plots in dense conifer stands, which were not pertinent for this analysis.

During the field season we periodically compared the number of plots sampled per stratum against the desired number of samples per stratum to ensure that sampled plots represented the correct balance of plots among strata.

Field Methods

We adapted rangeland sampling methods from Herrick and others (2005) for data collection in the John Day province in 2007. Locations of plot centers in a global positioning system (GPS) were pre-assigned from the sample point allocation process described above. A plot (and sample

unit) consisted of three 50-m transects radiating from a center stake. The three transects were arranged in a spoke design, with meter tapes laid out at azimuths of 0, 120, and 240 degrees from the plot center. The total area represented by each plot was thus ~0.8 ha. Line-intercept sampling was used along the three transects to assess canopy cover of trees and shrubs. The total length of canopy (in cm) intersecting the transect was recorded for each species. Intercept measures for dead trees and shrubs were also recorded, due to their importance in indicating current ecological function.

Tree density and size class were measured at each site using a modified Forest Inventory and Analysis plot design (USDA Forest Service 2007). Trees were measured in four circular, 7.3-m radius subplots. One subplot was located at the plot center; the other three were located at points 36.6 m from the plot center along each of the three transect lines.

Within each subplot, all trees were tallied by species. Junipers were assigned to one of four height classes: < 0.3 m, 0.3 – 1 m, 1 – 2 m, and > 2 m; and to one of two growth forms for trees > 2 m tall (representing trees from pre- or post-European settlement; pre-settlement trees are > 140 years old) (Gascho Landis and Bailey 2006; Miller and others 2005).

Table 1—Allocation of sampling points across sampling strata.

Stratum Number	Stratum Description ^a	Percent of Target Frame	Percent of Sampling Frame ^b	Projected Number of Sample Plots ^c	Actual Number of Plots Sampled
1	Non-sagebrush, far from juniper stands	46	68	7	4
2	Other sagebrush, far from juniper stands	15	9	71	51
3	Low/mountain big sagebrush, far from juniper stands	5	2	21	21
4	Non-sagebrush, near juniper stands	16	13	7	10
5	Other sagebrush, near juniper stands	10	4	57	54
6	Low/mountain big sagebrush, near juniper stands	6	3	29	26
7	Juniper stands	4	1	7	12
Total		100	100	200	178

^aFar from juniper stands = 1,600 – 5,000 m; near juniper stands = < 1,600 m.

^bFollowing application of filters to remove non-targeted or inaccessible areas.

^cFollowing application of weighting factors that incorporated the proportion of available pixels in a stratum and the estimated variance of juniper densities within that stratum.

RESULTS

We sampled trees and shrubs on 178 plots throughout the John Day province (table 1). Juniper trees were recorded on 150 (84 percent) of the sample plots; plots without juniper were found in all strata except 1 and 7 (table 2). Among the 2,254 trees counted, 2,225 (98.7 percent) were live. Dead western junipers were recorded in only 17 (9.6 percent) plots, and nearly all were in the “other sagebrush” strata (in other words, 2 and 5). Mean density of dead junipers across all plots and size/age classes was 2.4 trees/ha, compared to 186.7 trees/ha for live trees. Within the 17 plots with dead trees, most dead junipers were large (> 2 m), pre-settlement trees (mean density = 16.7 trees/ha); only 3 post-settlement dead junipers were recorded in the > 2 m size class.

We encountered pre-settlement trees in only 47 (26 percent) plots (table 2), and < 5 percent of the total junipers tallied were pre-settlement trees. The largest number and percentage of plots with pre-settlement juniper were in other sagebrush, far from juniper (stratum 2).

Juniper Density

Mean densities of pre-settlement trees ranged from 0 in stratum 3 to 17.6 trees/ha in stratum 2 (table 3). Notably, density of old trees in plots mapped as juniper (in other words, stratum 7) was only moderate (6.2 trees/ha). A high proportion of plots in all strata contained post-settlement junipers, with the exception of stratum 3 (table 2). Mean densities of post-settlement trees ranged from 74.7 to 211.3 trees/ha in non-woodland strata (in other words, plots within strata 1 to 6; $n = 166$) to 456.7 trees/ha in juniper plots ($n = 12$) (table 3). Juniper in non-woodland plots was most abundant in sites adjacent to juniper stands (in other words, plots within strata 4 to 6) vs. the equivalent strata far from juniper (in other words, strata 1 to 3), and in “other sagebrush” communities (in other words, plots within strata 2 and 5) compared to low/mountain big sagebrush (in other words, strata 3 and 6).

Densities of post-settlement trees were greatest in the > 2.0 -m size class (mean = 82.0 trees/ha), followed by the 0.3–1 m size class (mean = 51.7 trees/ha) (table 4). Mean density in the smallest size class (< 0.3 m tall) was 21.4 trees/ha. Comparing within strata, juniper densities were greatest in the > 2 m-tall size class for all strata except the two low/mountain big sagebrush strata (3 and 6), for which densities were greatest in the 0.3 – 1 m class.

Canopy Cover

Canopy cover of juniper, as estimated with line intercept, ranged from 0 ($n = 47$ plots; all strata except 7) to 36.4 percent within a juniper plot. Canopy cover of juniper trees was highest in stratum 7, as expected (figure 4). Outside this stratum (in other words, in sagebrush and other vegetation communities) juniper canopy cover was lowest in strata 3 and 6 (low/mountain big sagebrush), and highest in stratum 5, near existing juniper stands. Sagebrush canopy cover was

greatest in the two low/mountain big sagebrush strata, and lowest in the two non-sagebrush strata (1, and 4) and in juniper stands (figure 4). Canopy cover of all sagebrush taxa combined was negatively correlated with canopy cover of juniper in our plots ($r = -0.45$; figure 5).

DISCUSSION

Dynamics of juniper woodlands in the John Day province paralleled those recorded in several other studies conducted across the western United States (for example, Gedney and others 1999; Miller and others 2005, 2008), and suggest widespread encroachment of western juniper into sagebrush and other shrublands of central Oregon. Two of our sampling strata had juniper present on all plots, including juniper stands themselves and stratum 1 (non-sagebrush, far from juniper). We sampled only four plots in this stratum, and all were in mixed conifer forests, except for one juniper-encroached scabland sagebrush site.

We encountered few dead junipers, most of which were pre-settlement trees. The mean density of dead pre-settlement trees (in plots with dead trees present) was 16.8 trees/ha, similar to that reported for standing dead trees in old-growth juniper stands on aeolian soils of the High Desert province (up to 14.8 trees/ha; Miller and others 2005).

Six ecological provinces of the Intermountain West, including the John Day, support pre-settlement western juniper (Miller and others 1999b). Old junipers in the sedimentary soils of the John Day province have not been well-studied, but are typically widely spaced, resulting in very low densities with insufficient understory fuels to carry a fire (Miller and others 1999b). The low density of pre-settlement junipers that we recorded in juniper plots (6.7 trees/ha) confirms this pattern.

We found low percentages of mixed-age (in other words, co-occurrence of both pre- and post-settlement trees) plots and of old trees (compared to total trees) in the John Day province, suggesting that historically, junipers were widely scattered across the landscape. Our findings are similar to those reported in pinyon-juniper woodlands within four other ecological provinces in the Intermountain West (Miller and others 2008). The percentage of mixed-age plots in that study ranged from 16 percent to 30 percent in Nevada, Oregon, and Utah. Similar to our results, the percentage of old juniper trees, relative to total junipers, ranged from < 2 percent in Nevada, Oregon, and Utah sites to 10 percent in Idaho sites. A study of western juniper woodlands in four sites in Oregon and Idaho also reported low percentages (1 to 10 percent) of pre-settlement trees, and 17 percent of the juniper stands in Oregon were mixed age (Johnson and Miller 2008). The density of pre-settlement junipers we recorded was somewhat lower than that found by Miller and others (2008), who reported pre-settlement tree densities of 20 and 27 trees/ha in their Idaho sites, where old trees were most abundant.

Table 2—Presence of western juniper trees in plots sampled in the John Day province, central Oregon, 2007 ($n = 178$); data refer to live trees only. See text for definitions of pre- and post-settlement trees.

Stratum Number	Stratum Description ^a	No. Plots (% of total)	Plots with Pre- settlement Juniper (% of stratum)	Plots with Post- settlement Juniper (% of stratum)	Plots with Juniper Absent (% of stratum)
1	Non-sagebrush, far from juniper stands	4 (2.3)	1 (25.0)	4 (100)	0 (0.0)
2	Other sagebrush, far from juniper stands	51 (28.7)	25 (49.0)	46 (90.0)	5 (10.0)
3	Low/mountain big sagebrush, far from juniper stands	21 (11.8)	0 (0.0)	10 (47.6)	11 (52.3)
4	Non-sagebrush, near juniper stands	10 (5.6)	2 (20.0)	8 (80.0)	1 (10.0)
5	Other sagebrush, near juniper stands	54 (30.3)	14 (25.9)	48 (88.9)	6 (11.1)
6	Low/mountain big sagebrush, near juniper stands	26 (14.6)	3 (11.5)	21 (80.8)	5 (19.2)
7	Juniper stands	12 (6.7)	2 (16.7)	12 (100)	0 (0.0)
Total		178 (100)	47 (26.4)	149 (83.7)	28 (15.7)

^aFar from juniper stands = 1,600 – 5,000 m; near juniper stands = < 1,600 m.

Table 3—Mean density (number of trees/ha) and 90percent confidence interval (CI) for western juniper in plots sampled in the John Day province, central Oregon, 2007 ($n = 178$); data refer to live trees only. See text for definitions of pre- and post-settlement trees.

Stratum Number	Stratum Description ^a	All Juniper		Post-settlement Juniper		Pre-settlement Juniper	
		Mean	CI	Mean	CI	Mean	CI
1	Non-sagebrush, far from juniper stands	119.5	125.6	104.5	131.1	14.9	24.6
2	Other sagebrush, far from juniper stands	177.1	38.9	159.6	39.1	17.6	5.4
3	Low/mountain big sagebrush, far from juniper stands	74.7	63.8	74.7	63.8	0.0	NA
4	Non-sagebrush, near juniper stands	167.2	90.3	164.3	89.2	3.0	3.3
5	Other sagebrush, near juniper stands	220.4	41.0	211.3	40.1	9.1	5.0
6	Low/mountain big sagebrush, near juniper stands	116.0	66.8	113.7	66.1	2.3	2.2
7	Juniper stands	462.9	152.0	456.7	155.5	6.2	8.3

Table 4—Mean density (number of trees/ha) of post-settlement western juniper trees by height class in plots sampled in the John Day province, central Oregon, 2007 ($n = 178$). Data refer to live trees only; post-settlement trees are < 140 years old.

Stratum Number	Stratum Description ^a	< 0.3 m	0.3–1 m	1–2 m	> 2 m
1	Non-sagebrush, far from juniper stands	26.1	22.4	11.2	44.8
2	Other sagebrush, far from juniper stands	19.3	43.6	22.3	74.4
3	Low/mountain big sagebrush, far from juniper stands	8.5	39.1	10.7	16.4
4	Non-sagebrush, near juniper stands	29.9	38.8	13.4	82.1
5	Other sagebrush, near juniper stands	22.1	60.8	23.8	104.5
6	Low/mountain big sagebrush, near juniper stands	19.0	40.2	18.4	36.2
7	Juniper stands	46.0	112.0	58.5	240.2

Post-settlement junipers were far more abundant than old junipers in our sample plots. At least 80 percent of the plots in all strata except stratum 3 had post-settlement trees present. The density of post-settlement trees in juniper (in other words, stratum 7) plots (456.7 trees/ha) was comparable to that reported by Miller and others (2008) for pinyon-juniper woodlands of the Intermountain West, which ranged from 197 trees/ha in Idaho to 885 trees/ha in Utah. The majority of juniper forest in Oregon occurs in stands with < 123 trees/ha (Gedney and others 1999). Juniper densities exceeded this value in nearly 50 percent of our plots, most of which were classified by LANDFIRE not as juniper, but as sagebrush. Densities of juniper seedlings as low as 62/ha are considered indicative of continued woodland expansion (Azuma and others 2005); about one-third ($n = 47$) of the sagebrush plots we sampled, and 60 (33.7 percent) plots altogether, had densities of small (0 to 1 m tall) junipers exceeding this value.

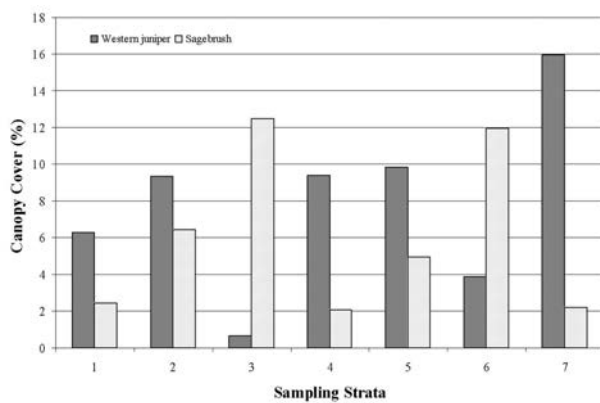


Figure 4—Canopy cover (percent) of western juniper trees and sagebrush in plots sampled in the John Day province, central Oregon, 2007 ($n = 178$). Strata are defined as follows: 1 = non-sagebrush, far from juniper stands, 2 = other sagebrush, far from juniper stands, 3 = low/mountain big sagebrush, far from juniper stands, 4 = non-sagebrush, near juniper stands, 5 = other sagebrush, near juniper stands, 6 = low/mountain big sagebrush, near juniper stands, and 7 = juniper stands. Far from juniper stands = 1,600 to 5,000 m; near juniper stands = < 1,600 m.

Comparison of densities of post-settlement junipers across size classes revealed that the dominant size class was > 2 m. However, junipers in the 0.3 to 1 m class were also abundant, suggesting continued establishment and encroachment of junipers in the John Day province. The juniper densities and associated sagebrush canopy cover we measured suggest active expansion of western juniper into shrublands of the John Day province, characterized as Phase I or II woodlands.⁴

Further exacerbating predicted juniper expansion in central Oregon are effects of increasing atmospheric CO₂ levels.

Knapp and others (2001), by using matched sets of drought and wet years pre- and post-1950, demonstrated consistently more rapid growth rates of western juniper in central Oregon in recent decades, presumably a fertilization effect of CO₂ that increases water use efficiency of juniper. The precise relation of increasing CO₂ levels and western juniper expansion is not well understood, however (Miller and others 2005; Soule and Knapp 1999).

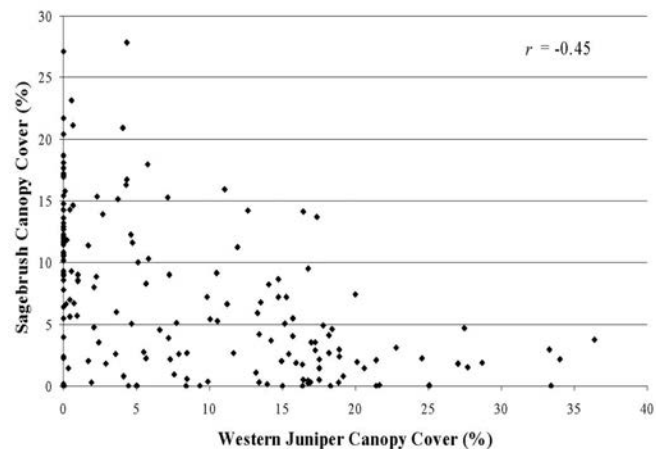


Figure 5—Canopy cover of western juniper trees and sagebrush (all taxa combined) on plots within seven strata sampled in the John Day province, central Oregon, 2007 ($n = 178$). (See table 1 for strata descriptions).

Densities of post-settlement junipers were consistently greater in plots located near existing juniper stands vs. plots in similar land-cover classes but farther from juniper (in other words, “near” vs. “far” strata). These results confirm the greater likelihood of juniper seed dispersal and subsequent establishment of trees within 1600 m of juniper stands. Birds are the most important disseminators of western juniper seeds, especially at local scales (Chambers and others 1999; Miller and others 2005). Although most seeds are deposited near the source tree, frugivorous mammals or flocks of birds such as American robins (*Turdus migratorius*), bluebirds (*Sialia* spp.), or waxwings (*Bombacilla* spp.) may disperse seeds several kilometers away (Chambers and others 1999). Increasing densities of seed-bearing trees, and thus seed availability, increase the likelihood of longer dispersal events (Chambers and others 1999).

⁴Phases of woodland development have been described as follows: Phase I, trees are present but shrubs and herbs are the dominant vegetation that influence ecological processes on the site; Phase II, trees are co-dominant with shrubs and herbs and all three vegetation layers influence ecological processes on the site; and Phase III, trees are the dominant vegetation and the primary plant layer influencing ecological processes on the site (from Miller and others 2008).

Multiple lines of evidence from our work suggest that low and mountain big sagebrush plots had the lowest levels of current juniper encroachment in the John Day province. This finding contrasts with work in the High Desert and Klamath provinces in southern Oregon and northern Nevada, where western juniper had encroached into mountain big sagebrush and low sagebrush communities more than any other sagebrush types (Miller and others 1999a). In our study, strata with the highest percentage of plots where juniper was absent were low and mountain big sagebrush sites, which typically occur at higher elevations that can be colder and/or more moist than is optimal for western juniper establishment (West and others 1978). These same environmental conditions in mountain big sagebrush sites also enhance restoration efforts; recovery in mountain big sagebrush is typically more rapid than in other sagebrush types following treatment to remove juniper (Miller and others 2005). Among our seven strata, low and mountain big sagebrush also had the lowest proportion of plots with pre-settlement junipers present, similar to results in studies in Oregon and Idaho (Johnson and Miller 2008). Miller and others (2005) noted the rarity of old trees or stumps in post-settlement juniper communities associated with mountain big sagebrush. Finally, canopy cover of juniper was lowest in the low and mountain big sagebrush strata.

In contrast to existing levels of encroachment, low and mountain big sagebrush sites may be at high risk of future woodland expansion in the John Day province. These two strata were the only ones in which juniper densities were greatest in the 0.3 to 1 m size class (table 4), suggesting relatively recent tree establishment but future growth and infilling. Johnson and Miller (2006) estimated a mean of 90 years for invaded mountain big sagebrush sites to reach Phase II. Historically, fire return intervals of < 50 years in mountain big sagebrush were probably adequate to suppress the encroachment of western juniper. However, sharp declines in fire frequency in mountain big sagebrush were reported beginning in the late 1800s, resulting in increased opportunities for western juniper establishment in these plant communities (Miller and others 2005). Thus, mountain big sagebrush may be highly susceptible to future encroachment of western juniper (Johnson and Miller 2006), especially in our study area.

Our finding of decreasing canopy cover of sagebrush with increasing canopy cover of juniper corroborates results of previous studies (Blackburn and Tueller 1970; Bunting and others 1999; Miller and others 2005; Tausch and Tueller 1990), although this relationship is not universal (for example, see Gedney and others 1999; Knapp and Soule 1998). As juniper crown cover increases, understory composition changes, with decreasing understory diversity and eventual loss of the shrub component (Bunting and others 1999; Gedney and others 1999; Miller and others

1999a). Sharp increases in shrub canopy cover also are seen following removal of juniper (Miller and others 2005).

The ubiquity of juniper in our sampling plots was not expected; 152 (85 percent) of our plots were mapped as sagebrush-dominated ecological systems by LANDFIRE. Yet in 53 (35 percent) of these plots, canopy cover of juniper exceeded 10 percent, the threshold for mapping juniper woodlands. These results, along with other explorations of land cover as mapped by LANDFIRE EVT in central Oregon, strongly suggest that western juniper is under-represented and sagebrush communities over-represented by LANDFIRE (data on file with the PNW Research Station, La Grande Forestry and Range Sciences Laboratory, La Grande, OR).

Most juniper in Oregon occurs on private lands (Gedney and others 1999), thus management of this resource by public agencies must be closely coordinated among all stakeholders to ensure that comprehensive treatment strategies are successful (Azuma and others 2005). Management actions for western juniper should be preceded by the setting of precise goals and objectives for the management program, such as describing the desired ecological conditions (Miller and others 2007). Other steps include: (1) identifying the ecological site (for example, soils, potential natural vegetation), (2) assessing current conditions on the site, (3) considering the landscape in which management is planned, and (4) selecting appropriate treatments, such as prescribed fire or mechanical removal (Miller and others 2007).

In summary, the densities and size distribution of juniper trees that we observed pose substantial risk to sagebrush communities in central Oregon. Questions remain about the extent of sagebrush that has been invaded recently by western juniper versus communities that historically supported woodlands. However, our findings suggest that within sagebrush communities of the John Day province, intensive management through removal of encroaching western juniper may be prudent, while retaining old-growth juniper stands.

ACKNOWLEDGMENTS

This work was supported by funding from the Western Wildland Environmental Threat Assessment Center (WWETAC), PNW Research Station, USDA Forest Service. Additional funding and in-kind support were provided by the USDA Forest Service, Pacific Northwest Research Station and National Forest System, Washington Office. We are especially grateful to Jerry Beatty and Charles ("Terry") Shaw of WWETAC for their continued support, and Judy Haigler for logistical help throughout the field season. John Swanson, on detail from the Forest Service to the BLM Prineville District Office, was instrumental in

establishing initial contacts for us with key personnel throughout the province. Our field crews benefited from the guidance and support of Ed Horn and Randy Hinton (BLM Prineville District). Jeff Rose (BLM Burns District) helped with logistics in that portion of our study area. We thank Rick Miller for spending time in the field to help our crews with identification of juniper phases. Our field crew leader, Kent Coe, and technicians Dan Barrett and Chris Mongeon, worked diligently to collect data across central Oregon. Jennifer Boyd created maps, analyzed data, and provided all-around support on the project. Bridgett Naylor assisted with field sampling logistics and GIS analysis. Assistance with sampling design and other aspects of the project was provided by our collaborators Bryan Endress and Catherine Parks. Ken Brewer, Mark Finco, and Denise Laes of the Remote Sensing Applications Center (Salt Lake City, Utah) helped with designing our strata for point allocation, obtaining aerial photography in the province, and conducting crosswalks of land-cover maps.

REFERENCES

- Anderson, E.W., Borman, M.M., Krueger, W.C. 1998. Ecological provinces of Oregon: a treatise on the basic ecological geography of the state. Corvallis, OR: Oregon State University, Oregon Agricultural Experiment Station. 139 p.
- Azuma, D.L., Hiserote, B.A., Dunham, P.A. 2005. The western juniper resource of eastern Oregon, 1999. Resour. Bull. PNW-RB-249. Portland, OR: USDA, Forest Service, Pacific Northwest Research Station. 18 p.
- Bates, J.D., Miller, R.F., Svejcar, T.J. 2000. Understory vegetation response following cutting of western juniper. *Journal of Range Management*. 53: 119–126.
- Belsky, A.J. 1996. Viewpoint: western juniper expansion: is it a threat to arid northwestern ecosystems? *Journal of Range Management*. 49: 53–59.
- Betancourt, J.L. 1987. Paleoecology of pinyon–juniper woodlands: summary. In: Everett, R.L., comp. Proceedings: pinyon–juniper conference. Gen. Tech. Rep. INT-215. Ogden, UT: USDA, Forest Service, Intermountain Research Station: 129–139.
- Blackburn, W.H., Tueller, P.T. 1970. Pinyon and juniper invasion in black sagebrush communities in east-central Nevada. *Ecology*. 51: 841–848.
- Brown, J.K., Smith, J.K., eds. 2000. Wildland fire in ecosystems: effects of fire on flora. Gen. Tech. Rep. RMRS-42, vol. 2. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station. 257 p.
- Bunting, S.C., Kingery, J.L., Hemstrom, M.A. [and others]. 2002. Altered rangeland ecosystems in the interior Columbia Basin. Gen. Tech. Rep. PNW-553. Portland, OR: Pacific Northwest Research Station. 71 p.
- Bunting, S.C., Kingery, J.L., Strand, E. 1999. Effects of succession on species richness of the western juniper woodland/sagebrush steppe mosaic. In: Monsen, S.B., Stevens, R., comps. Proceedings: ecology and management of pinyon-juniper communities within the interior west. Proceedings RMRS-P-9. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station: 76–81.
- Campbell, S., Azuma, D., Weyermann, D. 2003. Forests of eastern Oregon: an overview. Gen. Tech. Rep. PNW-578. Portland, OR: USDA, Forest Service, Pacific Northwest Research Station. 31 p.
- Chambers, J.C. 2001. *Pinus monophylla* establishment in an expanding piñon-juniper woodland: environmental conditions, facilitation, and interacting factors. *Journal of Vegetation Science*. 12: 27–40.
- Chambers, J.C., McArthur, E.D., Monson, S.B. [and others]. 2005. Sagebrush steppe and pinyon-juniper ecosystems: effects of changing fire regimes, increased fuel loads, and invasive species. Project #00-1-1-03 Joint Fire Sciences Report. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station. 66 p.
- Chambers, J.C., Schupp, E.W., Vander Wall, S.B. 1999. Seed dispersal and seedling establishment of piñon and juniper species within the piñon-juniper woodland. In: Monsen, S.B., Stevens, R., comps. Proceedings: ecology and management of pinyon-juniper communities within the interior west. Proceedings RMRS-P-9. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station: 29–34.
- Clark, J.S., Fastie, C., Hurtt, G. [and others]. 1998. Reid's paradox of rapid plant migration. *BioScience*. 48: 13–24.
- Comer, P., Faber-Langendoen, D., Evans, R. [and others]. 2003. Ecological systems of the United States: a working classification of U.S. terrestrial systems. Arlington, Va.: NatureServe. 75 p.
- Connelly, J.W., Knick, S.T., Schroeder, M.A., Stiver, S.J. 2004. Conservation assessment of greater sage-grouse and sagebrush habitats. Cheyenne, WY: Western Association of Fish and Wildlife Agencies. [not paged].
- Coppedge, B.R., Engle, D.M., Masters, R.E., Gregory, M.S. 2004. Predicting juniper encroachment and CRP effects on avian community dynamics in southern mixed-grass prairie, USA. *Biological Conservation*. 115: 431–441.
- Dobkin, D.S., Sauder, J.D. 2004. Shrubsteppe landscapes in jeopardy: distributions, abundances, and the uncertain future of birds and small mammals in the intermountain west. Bend, OR: High Desert Ecological Research Institute. 199 p.
- Ferguson, S.A. 2001. Climatic variability in eastern Oregon and Washington. *Northwest Science*. 75(Special Issue): 62–69.
- Franklin, J.F., Dyrness, C.T. 1973. Natural vegetation of Oregon and Washington. Gen. Tech. Rep. PNW-8. Portland, OR: USDA, Forest Service, Pacific Northwest Forest and Range Experiment Station. 417 p.
- Gascho Landis, A.M., Bailey, J.D. 2006. Predicting age of pinyon and juniper trees using allometric relationships. *Western Journal of Applied Forestry*. 21: 203–206.
- Gedney, D.R., Azuma, D.L., Bolsinger, C.L., McKay, N. 1999. Western juniper in eastern Oregon. Gen. Tech. Rep. PNW-464. Portland, OR: USDA, Forest Service, Pacific Northwest Research Station. 53 p.
- Gholz, H.L. 1980. Structure and productivity of *Juniperus occidentalis* in central Oregon USA, *American Midland Naturalist*. 103: 251–261.
- Herrick, J.E., Van Zee, J.W., Havstad, K.M. [and others]. 2005. Monitoring manual for grassland, shrubland, and savanna ecosystems. Tucson, AZ: University of Arizona Press. 236 p.
- Hubbard, K.A., Schimel, D.S., Archer, S., Ojima, D.S., Parton, W. 2003. Grassland to woodland transitions: integrating changes in landscape structure and biogeochemistry. *Ecological Applications*. 13: 911–926.
- Huxman, T.E., Wilcox, B.P., Breshears, D.D. [and others]. 2005. Ecohydrological implications of woody plant encroachment. *Ecology*. 86: 308–319.
- Johnsen, T.N. 1962. One-seed juniper invasion of northern Arizona grasslands. *Ecological Monographs*. 32: 187–207.
- Johnson, D.D., Miller, R.F. 2006. Structure and development of expanding western juniper woodlands as influenced by two topographic variables. *Forest Ecology Management*. 299: 7–15.
- Johnson, D.D., Miller, R.F. 2008. Intermountain presettlement juniper: distribution, abundance, and influence on postsettlement expansion. *Journal of Range Management*. 61:82–92.
- Knapp, P.A., Soulé, P.T. 1998. Recent *Juniperus occidentalis* (western juniper) expansion on a protected site in central Oregon. *Global Change Biology*. 4: 347–357.
- Knapp, P.A., Soulé, P.T., Grissino-Mayer, H.D. 2001. Detecting regional effects of increased atmospheric CO₂ on growth rates of western juniper. *Global Change Biology*. 7: 903–917.
- Knick, S.T. 1999. Requiem for a sagebrush ecosystem? *Northwest Science*. 73: 53–57.
- Knick, S.T., Dobkin, D.S., Rotenberry, J.T. [and others]. 2003. Teetering on the edge or too late? Conservation and research issues for avifauna of sagebrush habitats. *Condor*. 105: 611–634.
- Miller, R., Svejcar, T., Rose, J. 1999a. Conversion of shrub steppe to juniper woodland. In: Monsen, S.B., Stevens, R., comps. Proceedings: ecology and management of pinyon-juniper communities within the interior west. Proceedings RMRS-P-9. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station: 385–390.
- Miller, R., Svejcar, T., Rose, J. 2000. Impacts of western juniper on plant community composition and structure. *Journal of Range Management*. 53: 574–585.
- Miller, R., Tausch, R., Waichler, W. 1999b. Old-growth juniper and pinyon woodlands. In: Monsen, S.B., Stevens, R., comps. Proceedings: ecology and management of pinyon-juniper communities within the interior west. Proceedings RMRS-P-9. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station: 375–384.

- Miller, R.F., Bates, J.D., Svejcar, T.J. [and others]. 2005. Biology, ecology, and management of western juniper. Tech. Bull. 152. Corvallis, OR: Oregon State University, Agricultural Experiment Station. 77 p.
- Miller, R.F., Bates, J.D., Svejcar, T.J. [and others]. 2007. Western juniper field guide: asking the right questions to select appropriate management actions. USDI, Geological Survey, Reston, VA. Circular 1321. 61 p.
- Miller, R.F., Eddleman, L.L. 2000. Spatial and temporal changes of sage-grouse habitat in the sagebrush biome. Tech. Bull. 151. Corvallis, OR: Oregon State University, Agricultural Experiment Station. 35 p.
- Miller, R.F., Rose, J.A. 1995. Historic expansion of *Juniperus occidentalis* (western juniper) in southeastern Oregon. Great Basin Naturalist. 55: 37–45.
- Miller, R.F., Tausch, R.J. 2001. The role of fire in pinyon and juniper woodlands: a descriptive analysis. In: Galley, K.E.M., Wilson, T.P., eds. Proceedings of the invasive species workshop: the role of fire in the control and spread of invasive species. Tall Timbers Research Station Misc. Pub. 11. Tallahassee, FL: Tall Timbers Research Station: 15–30.
- Miller, R.F., Tausch, R.J., McArthur, E.D. [and others]. 2008. Age structure and expansion of piñon-juniper woodlands: a regional perspective in the Intermountain West. Research Paper Report RMRS-RP-69. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station. 15 p.
- Miller, R.F., Wigand, P.E. 1994. Holocene changes in semi-arid pinyon-juniper woodlands: response to climate, fire, and human activities in the U.S. Great Basin. BioScience 44: 465–474.
- Mock, C.J. 1996. Climatic controls and spatial variations of precipitation in the western United States. Journal of Climate. 9: 1111–1125.
- Nelson, D.R., Fairchild, J.A., Nunn-Hatfield, C.R. 1999. Political guidelines for management and restoration of pinyon and juniper woodlands. In: Monsen, S.B., Stevens, R., comps. Proceedings: ecology and management of pinyon-juniper communities within the interior west. Proceedings RMRS-P-9. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station: 371–374.
- Pavlacky, D.C., Jr., Anderson, S.H. 2001. Habitat preferences of pinyon-juniper specialists near the limit of their geographic range. Condor. 103: 322–331.
- Pieper, R.D., Tanaka, J., Harp, A. J. [and others]. 2002. Juniper and piñon-juniper woodlands in the west: can the system support biomass removal for energy production? Caldwell, ID: Policy Analysis Center for Western Public Lands, Univ. of Idaho. 33 p.
- Reinkensmeyer, D.P., Miller, R.F., Anthony, R.G. [and others]. 2008. Winter and early spring bird communities in grasslands, shrubsteppe, and juniper woodlands in central Oregon. Western North American Naturalist. 68: 25–35.
- Rollins, M.G., Frame, C.K., tech. eds. 2006. The LANDFIRE Prototype Project: nationally consistent and locally relevant geospatial data for wildland fire management. Gen. Tech. Rep. RMRS-175. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station. 416 p.
- Rowland, M.M., Suring, L.H., Wisdom, M.J. [and others]. 2005. Habitats for vertebrate species of conservation concern. In: Wisdom, M.J., Rowland, M.M., and Suring, L.H., eds. Habitat threats in the sagebrush ecosystem: methods of regional assessment and applications in the Great Basin. Lawrence, KS: Alliance Communications Group: 163–204.
- Rumble, M.A., Gobeille, J.E. 1995. Wildlife associations in Rocky Mountain juniper in the northern Great Plains, South Dakota. In: Shaw, D.W., Aldon, E.F., LoSapio C., eds. Desired future conditions for piñon-juniper ecosystems. Gen. Tech. Rep. RM-258. Fort Collins, CO: Forest Service, Rocky Mountain Forest and Range Experiment Station: 80–90.
- Schaefer, R.J., Thayer, D.J., Burton, T.S. 2003. Forty-one years of vegetation change on permanent transects in northeastern California: implications for wildlife. California Fish and Game. 89:5 5–71.
- Schupp, E.W., Chambers, J.C., Vander Wall, S.B. [and others]. 1999. Piñon and juniper seed dispersal and seedling recruitment at woodland ecotones. In: McArthur, E.D., Ostler, W.K., Wambolt, C.L., comps. Proceedings: shrubland ecotones. Proceedings RMRS-P-11. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station: 66–70.
- Soulé, P.T., Knapp, P.A. 1999. Western juniper expansion on adjacent disturbed near-relict sites. Journal of Range Management. 52: 525–533.
- Soulé, P.T., Knapp, P.A., Grissino-Mayer, H.D. 2003. Comparative rates of western juniper afforestation in south-central Oregon and the role of anthropogenic disturbance. The Professional Geographer. 55: 43–55.
- Suring, L.H., Wisdom, M.J., Tausch, R.J. [and others]. 2005. Modeling threats to sagebrush and other shrubland communities. In: Wisdom, M.J., Rowland, M.M., and Suring, L.H., eds. Habitat threats in the sagebrush ecosystem: methods of regional assessment and applications in the Great Basin. Lawrence, KS: Alliance Communications Group: 114–149.
- Tausch, R.J. 1999. Historic pinyon and juniper woodland development. In: Monsen, S.B., Stevens, R., comps. Proceedings: ecology and management of pinyon-juniper communities within the interior west. Proceedings RMRS-P-9. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station: 12–19.
- Tausch, R.J., Tueller, P.T. 1990. Foliage biomass and cover relationships between tree- and shrub-dominated communities in pinyon-juniper woodlands. Great Basin Naturalist. 50: 121–134.
- Taylor, C.A. 2007. Role of summer prescribed fire to manage shrub-invaded grasslands. In: Sosebee, R.E., Wester, D.B., Britton, C.M., McArthur, E.D., Kitchen, S.G., comps. Proceedings: shrubland dynamics – fire and water. Proceedings RMRS-P-47. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station: 52–55.
- USDA Forest Service. 2007. Forest inventory and analysis National core field guide. Volume 1: field data collection procedures for phase 2 plots. Version 4.0. Washington, DC: USDA, Forest Service, Forest Inventory and Analysis. 224 p.
- USDA Natural Resources Conservation Service. 2008. The PLANTS Database. (<http://plants.usda.gov>). Baton Rouge, LA: USDA, Natural Resources Conservation Service.
- USDI Fish and Wildlife Service. 2005. Endangered and threatened wildlife and plants, 12-month finding for petitions to list greater sage-grouse as threatened/endangered, proposed rule. Federal Register. 70: 2244–2282.
- Vander Wall, S.B., Balda, R.P. 1977. Coadaptations of the Clark's nutcracker and the piñon pine for efficient seed harvest and dispersal. Ecological Monographs 47:89–111.
- Vander Wall, S.B., Balda, R.P. 1981. Ecology and evolution of food storage behavior in conifer-seed-caching corvids. Zeitschrift für Tierpsychol. 56:217–242.
- West, N.E., Tausch, R.J., Rea, K.H., Tueller, P.T. 1978. Phytogeographic variation within juniper-pinyon woodlands of the Great Basin. Great Basin Naturalist Memoirs. 2: 119–136.
- West, N.E., Tausch, R.J., Tueller, P.T. 1998. A management-oriented classification of pinyon-juniper woodlands of the Great Basin. Gen. Tech. Rep. RMRS-12. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station. 42 p.
- West, N.E., Van Pelt, N.S. 1987. Successional patterns in pinyon–juniper woodlands. In: Everett, R.L., comp. Proceedings: pinyon–juniper conference. Gen. Tech. Rep. INT-215. Ogden, UT: USDA, Forest Service, Intermountain Research Station: 43–52.
- Wisdom, M.J., Rowland, M.M., and Suring, L.H., eds. 2005. Habitat threats in the sagebrush ecosystem: methods of regional assessment and applications in the Great Basin. Lawrence, KS: Alliance Communications Group.

Relationship of Wyoming Big Sagebrush Cover to Herbaceous Vegetation

Bok F. Sowell¹, Carl L. Wambolt², Jennifer K. Woodward², and Vanessa R. Lane²

ABSTRACT

We measured 328 sites in northern, central, and southern Montana and northern Wyoming during 2003 to test the relationship of herbaceous cover to Wyoming big sagebrush (*Artemisia tridentata wyomingensis*) cover. Long term annual precipitation at all sites was approximately 31 cm. Sagebrush and total herbaceous cover varied from 5 to 45 percent and 3.5 to 55 percent, respectively. Simple linear regression was the best fit model for predicting herbaceous cover from sagebrush cover using the highest R_a^2 values as the model selection criteria. In northern Montana, herbaceous vegetation was predicted by sagebrush cover with the following model: $Y = 37.4 - 0.61X$ ($R_a^2 = 0.16$, $P < 0.001$, $n = 87$). In central Montana, the model was $Y = 14.0 - 0.00X$ ($R_a^2 = 0.00$, $P = 1.0$, $n = 155$). In southern Montana, the model was $Y = 35.9 - 0.39X$ ($R_a^2 = 0.14$, $P < 0.001$, $n = 86$). When all sites were combined, the best fit model was $Y = 23.7 - 0.15X$ ($R_a^2 = 0.01$, $P < 0.061$, $n = 328$). This analysis determined that only 1 percent of the variation in herbaceous vegetation cover was associated with Wyoming big sagebrush cover. Management suggestions to reduce Wyoming big sagebrush in order to increase herbaceous production for greater sage-grouse (*Centrocercus urophasianus*) or livestock do not appear to be biologically sound. Keywords: *Artemisia tridentata wyomingensis*, line intercept, grass cover, *Centrocercus urophasianus*, forb cover, greater sage-grouse, sage-grouse habitat.

INTRODUCTION

It has been suggested that dense sagebrush (*Artemisia*) cover lowers greater sage-grouse (*Centrocercus urophasianus*) habitat quality and biological diversity (SRM 2006). The same publication states that sagebrush control can be used to enhance sage-grouse habitat by reducing sagebrush cover, which limits understory grass and forb production. Wright and Bailey (1982) recommended that removal of tall, thick sagebrush would release grasses and forbs from competition and result in increased yields for livestock grazing. In contrast, Miller and Eddleman (2001) concluded that there was little evidence that fire could be used to enhance sage-grouse habitat where there was a balance of native shrubs and perennial grasses and forbs. This statement implies there are conditions where shrubs and herbaceous vegetation were “unbalanced” and therefore might be manipulated. Welch and Criddle (2003) examined the relationship

between big sage cover and percent bare ground from several data sets and concluded that the calculated R^2 values averaging 0.05 for mountain big sagebrush (*A. tridentata vaseyana*) and Wyoming big sagebrush (*A. t. wyomingensis*) cover and perennial herbaceous cover were not significant. We modeled the data from 328 sites to determine the best-fit relationship between Wyoming big sagebrush cover and herbaceous cover.

MATERIALS AND METHODS

Study Sites

Our three study areas were separated by a total of approximately 355 km from the northern most to the southern most locations sampled. Northern Montana samples were taken approximately 80 km south of Malta in southern Phillips County. Approximately 60 percent of this area is publicly owned by the U.S. Bureau of Land Management (BLM), the U.S. Fish and Wildlife Service (FWS), and the state of Montana (Moynahan 2004). Annual precipitation averaged 31 cm with peak precipitation between April and July (WRCC 2004). Soil taxonomic units that characterized this area included Absher, Elloam, and Thoeny (USDA 1981). Elevation varied from 600 – 1,060 m. Wyoming big sagebrush was the dominant shrub at all sample locations. Plains silver sagebrush (*Artemisia cana*), greasewood (*Sarcobatus vermiculatus*), and rubber rabbitbrush (*Chrysothamnus nauseosus*) were relatively common at many sites. Western wheatgrass (*Pascopyron smithii*), and blue grama (*Bouteloua gracilis*) were the common dominant grasses, while Sandberg bluegrass (*Poa secunda*), needle-and-thread (*Hesperostipa comata*), and threadleaf sedge (*Carex filifolia*) were measured frequently. American vetch (*Vicia americana*), scarlet globemallow (*Sphaeralcea coccinea*), yellow sweetclover (*Melilotus officinalis*), and dandelion (*Taraxacum officinale*) were the most common forbs. Fringed sagewort (*Artemisia frigida*), clubmoss (*Selaginella densa*), and prickly pear (*Opuntia polyacantha*) were also common.

Measurements in the central Montana counties of Golden Valley and Musselshell were taken in an area centered about 30 km northwest of Roundup, Montana. Precipitation in this area averages 31 cm annually (NOAA 2004). Soil taxonomic units common to this area include Abor, Neldore, and Vanda (USDA 2003). Elevation is 826-1495 m. Wyoming big sagebrush was the dominant shrub although greasewood and plains silver sagebrush were also present. Western wheatgrass, Sandberg bluegrass, and blue grama were the dominant grasses, while green needlegrass (*Nassella viridula*), needle-and-thread, and threadleaf sedge

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Bok F. Sowell, Department of Animal and Range Sciences, Montana State University, Bozeman, Email: bok@montana.edu

²Carl L. Wambolt, Professors, Jennifer K. Woodward and Vanessa R. Lane, graduate research assistants, Department of Animal and Range Sciences, Montana State University, Bozeman, MT

were also common. Scarlet globemallow, wild onion (*Allium*), Hood's phlox (*Phlox hoodii*), and American vetch were the most abundant forbs.

Sampled sites in Bighorn County in southern Montana and adjacent Campbell County, Wyoming were mostly on private rangeland with some BLM and state land. Annual precipitation averaged 31 cm with peak precipitation occurring from April to June (NOAA 2003). Soil taxonomic units which characterize this area include Midway, Pierre, and Thedallun (USDA NRCS 2004). The elevation range is 762–1,314 m. Wyoming big sagebrush was the dominant shrub, although plains silver sagebrush, skunkbrush sumac (*Rhus aromatica*), common juniper (*Juniperus communis*) and rubber rabbitbrush were also present. Sandberg bluegrass, western wheatgrass, and Japanese brome (*Bromus japonicus*) were the dominant grasses, although green needlegrass, prairie junegrass (*Koeleria macrantha*), and bluebunch wheatgrass (*Pseudoroegneria spicata*) were also common. Desert alyssum (*Alyssum desertorum*), Hood's phlox, scarlet globemallow, American vetch, dandelion, and western yarrow (*Achillea millefolium*) were the most common forbs.

Sampling and Analysis

All sampling was conducted during late spring and early summer of 2003. Sagebrush cover was measured with the line-intercept method (Canfield 1941) using the procedures recommended as a standard and discussed by Connelly and others (2003) and Wambolt and others (2006). Line-intercepts were measured on two perpendicular 30 m N-S and E-W oriented transects. We measured sagebrush intercept in 3 cm units. We recorded openings in live foliage > 3 cm as nonsagebrush intercepts. Thus, we excluded both open spaces and dead portions of the plant. We obtained all measurements by vertically projecting a plumb bob from the transect line to plants to determine what intercepts from ground level to crown were directly beneath the line.

Herbaceous understory cover was measured using twenty 20 x 50 cm quadrats (Daubenmire 1959). The same N-S, E-W transects used to measure sagebrush cover were used to locate the quadrats for herbaceous sampling at 3 m intervals.

A number of linear and non-linear models were tested to determine the "best fit" relationship between sagebrush cover and herbaceous cover. Linear, quadratic, cubic, inverse, first order, sigmoidal, exponential decay, exponential growth and polynomial models were tested using Sigma Plot (2004). Maximum R_a^2 values were used to determine which model best fit our data.

RESULTS AND DISCUSSION

Simple linear regression models for all data sets resulted in the maximum R_a^2 values. In northern Montana, as sagebrush cover increased, herbaceous cover decreased ($R_a^2 = 0.16$, $P < 0.001$, $n = 87$), but only 16 percent of the variation in herbaceous cover was accounted for with change in sagebrush cover (figure.1). Although the regression is significant, this relationship indicates that if sagebrush cover at 20 percent were reduced to 5 percent, herbaceous cover would only be increased from 25 to 35 percent.

In central Montana, we found no relationship ($R_a^2 = 0.00$, $P = 1.0$, $n = 155$) between Wyoming big sagebrush cover and herbaceous cover (figure 1). We conclude that in this region, herbaceous cover is not influenced by Wyoming big sagebrush cover.

In southern Montana, herbaceous cover declined as sagebrush cover increased, but, 86 percent of the variation in herbaceous cover was not accounted for by sagebrush cover ($R_a^2 = 0.14$, $P < 0.001$, $n = 86$) (figure 1). Thus, if a site with 20 percent sagebrush cover was treated to reduce this cover to 5 percent, herbaceous cover would be expected to increase only 6 percent.

When all 328 sites were combined, a simple linear equation best explained the relationship between Wyoming big sagebrush cover and herbaceous cover (figure 1). There is a slight inverse relationship between the two variables (figure 1), but the $R_a^2 = 0.01$, ($P < 0.06$) indicates that across Montana and adjacent Wyoming, the variability in herbaceous cover cannot be explained by the amount of sagebrush cover. The model for all sites ($n = 328$) determined if an area had 20 percent sagebrush cover, the average herbaceous cover would be 20 percent. Thus, if sagebrush cover were reduced to 5 percent, our model predicts the herbaceous cover would only increase to 23 percent.

Some authors (Baxter 1996; SRM 2006) indicated that when big sagebrush cover increases over 12 to 15 percent, the understory production of other plants decreases as cover increases. However, Welch and Criddle (2003) found no significant relationship between sagebrush cover and bare ground ($R^2 = 0.0003$), which supports our conclusions with data from other regions. An additional concern arises from the fact that Baxter (1996) and SRM (2006) do not clarify the details of their sagebrush cover measurements and, therefore, it is possible, if not probable, that their 12 to 15 percent sagebrush cover would be much less using our methodology (Wambolt and others 2006).

Moynahan (2004) examined grass and forb cover from 2001 to 2003 in our northern Montana study area as part of a sage-grouse brood survival study. Perennial grass cover

did not change appreciably in three years, but forb cover nearly doubled, largely due to the biennial yellow sweetclover responding to favorable growing conditions. Brood survival rates increased 3.5 fold from 2001 (drought year) to 2003 likely due to the increased abundance of herbaceous cover. Moynahan's (2004) findings in combination with our results indicate that precipitation has a much greater influence on herbaceous cover than does the amount of sagebrush cover. Welch and Criddle (2003) also concluded that precipitation, plant species, and soil properties influenced ground and perennial grass cover more than sagebrush cover.

MANAGEMENT IMPLICATIONS

It has been suggested that sagebrush control can be used to enhance herbaceous vegetation (SRM 2006). We

determined that across our 328 study sites that a weak linear relationship exists between Wyoming big sagebrush cover and herbaceous cover ($Y = 23.7 - 0.15X$). However, 99 percent of the variation in herbaceous cover (3.5 to 55 percent) is not accounted for by changes in sagebrush cover (5 to 45 percent) alone ($R_a^2 = 0.01$, $P < 0.061$). Our data indicate a large majority of sites would fail to respond to sagebrush treatments as predicted with conventional wisdom (SRM 2006). Prescribed burning, prescribed grazing, herbicides and mechanical treatments have all been advocated to improve sage-grouse habitat where it is hypothesized that dense sagebrush cover limits herbaceous biomass (SRM 2006). Removing Wyoming big sagebrush cover to increase herbaceous vegetation for any purpose, including enhancing sage-grouse brood survival, does not appear to be biologically sound.

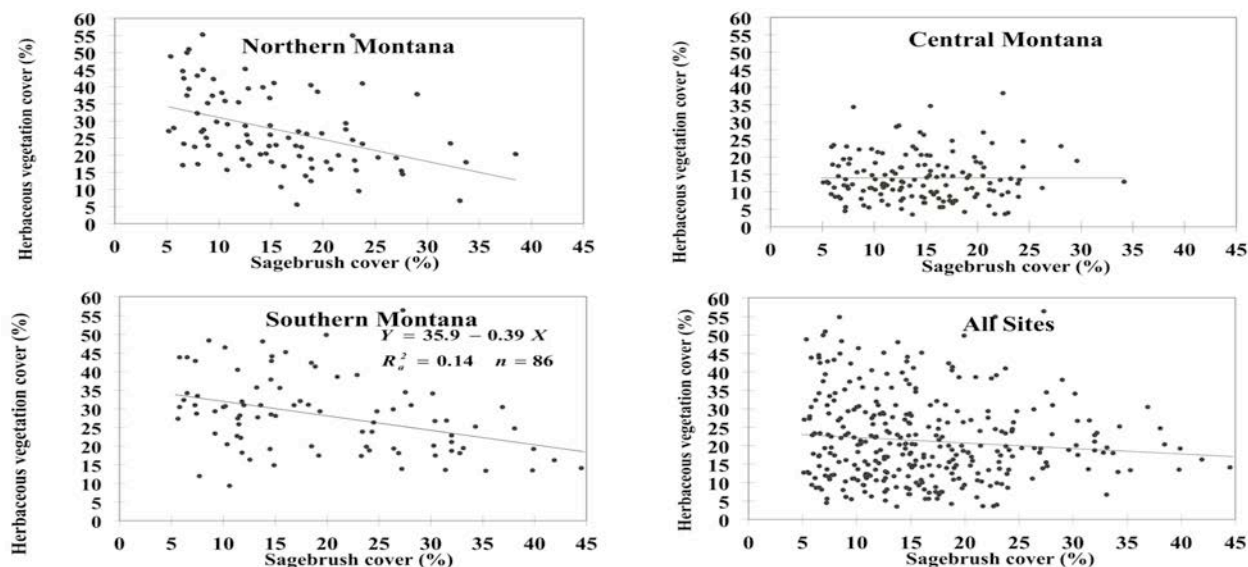


Figure 1—Relationship of Wyoming big sagebrush cover to herbaceous cover in northern, central, and southern Montana and over all sites. Funding and land for this research was provided by the Bureau of Land Management field offices in Montana, Montana Fish, Wildlife and Parks, and the Montana Agricultural Experiment Station.

REFERENCES

- Baxter, G. 1996. Improving rangeland health by thinning dense sagebrush stands with tebuthiuron (spike 20P). In: Evans, K.E., comp. Sharing common ground on western rangelands: proceedings of a livestock/big game symposium; 1996 Feb 26-28; Sparks, NV. INT-GTR-343. Ogden, UT: Intermountain Research Station: 60-62.
- Canfield, R.H. 1941. Application of the line interception method in sampling range vegetation. *Journal of Forestry* 39: 388-394.
- Connelly, J.W., Reese, K.P., Schroeder, M.A. 2003. Monitoring of greater sage-grouse habitats and populations. University of Idaho, College of Natural Resources Experiment Station Bulletin 80, Moscow.
- Daubenmire, R.F. 1959. A coverage method of vegetation analysis. *Northwest Science* 33: 43-64.
- Miller, R.F., Eddleman, L.L. 2001. Spatial and Temporal Changes of Sage Grouse Habitat in the Sagebrush Biome. Oregon State University Agricultural Experiment Station Technical Bulletin 151. Corvallis.
- Moynahan, B.J. 2004. Landscape-scale factors affecting population dynamics of greater sage-grouse in north-central Montana, 2001-2004. [Thesis]. University of Montana, Missoula.
- NOAA. 2003. [Online] *Annual climatological summary, Decker, Montana*. Available: <http://cdo.ncdc.noaa.gov/ancsum/ACS>.
- NOAA. 2004. [Online] *Annual climatological summary, Roundup, Montana*. Available: <http://cdo.ncdc.noaa.gov/ancsum/ACS>.
- Sigma Plot 6.0. 2004. Users Guide. SPSS Inc. 233 South Wacker Drive, 11th Floor, Chicago, IL 60606-6307.
- SRM. 2006 Ecology and Management of Sage-Grouse and Sage- Grouse Habitat, An issue paper created by the Society for Range Management. 10030 W 27th Ave. Wheat Ridge, CO 80215.
- USDA. 2003. Soil survey of Musselshell County, Montana. NRCS office, Bozeman, Montana.
- USDA. 1981. Technical guide. Section II-E-8. USDA-SCS-MT.
- USDA NRCS. 2004. [Online] *Soil data mart*. Available: <http://soildatamart.nrcs.usda.gov>.
- Wambolt, C.L., Frisina, M.R., Knapp, S.J., Frisina, R.M. 2006. Effect of method, site, and taxon on line-intercept estimates of sagebrush cover. *Wildlife Society Bulletin*. 34:440-445.
- Welch, B.L., Criddle, C. 2003. Countering Misinformation Concerning Big Sagebrush. Research Paper RMRS-RP-40. Ogden, UT: USDA, Forest Service, Rocky Mountain Research Station. 28p.
- WRCC. 2004. [Online] *Malta 35S, Montana climate summary*. Available: <http://www.wrcc.dri.edu/summary/climsmmt.html>.
- Wright, H.A., Bailey, A.W. 1982. Fire Ecology, United States and Southern Canada. John Wiley & Sons, New York.

Vegetation Dynamics at a Mojave Desert Restoration Site, 1992 to 2007

Jeffrey E. Ott¹, E. Durant McArthur², and Stewart C. Sanderson³

ABSTRACT

*The Twist Hollow restoration site on BLM land near St. George, Utah, had been badly disturbed by sand mining, rock quarrying, dumping, off-road vehicles and target shooting prior to its closure and treatment. In December 1992 the site was sculpted and drill seeded with Indian ricegrass (*Stipa hymenoides*), sand dropseed (*Sporobolus cryptandrus*), galleta (*Hilaria jamesii*), gooseberryleaf globemallow (*Sphaeralcea grossulariifolia*) Palmer penstemon (*Penstemon palmeri*), fourwing saltbush (*Atriplex canescens*) and winterfat (*Krascheninnikovia lanata*) in an effort restore natural vegetation and desert tortoise habitat. Vegetation was sampled before and after treatment and subsequently monitored over a period of 14 years. With the exception of winterfat, all the seeded species established and increased in density and/or cover during the period 1993-1998. Cheatgrass (*Bromus tectorum*) also increased and became dominant during this period. Seeding was most successful on a rockier substrate where fourwing saltbush became a dominant shrub. Drought conditions after 1998 corresponded with declines in most seeded species while broom snakeweed (*Gutierrezia sarothrae*), hairy goldenaster (*Chrysopsis villosa*), desert globemallow (*Sphaeralcea ambigua*) and sandsage (*Artemisia filifolia*) increased through recruitment from surrounding vegetation. Sandsage was the dominant shrub of nearby undisturbed sandy sites and grew rapidly following establishment in a sandier portion of the treated area. Cheatgrass and other annuals fluctuated from year to year in the treated area but had lower density and cover than nearby untreated areas throughout the monitoring period. We conclude that the restoration project's objectives have been met to varying degrees despite the limited persistence of the seeded species. Further research into management techniques aimed at reducing annual grasses and enhancing high-quality desert tortoise forage is recommended.*

INTRODUCTION

The Red Cliffs area near St. George, Washington County, Utah, is valued for its striking scenery and its ecological significance as part of the Mojave Desert ecosystem providing habitat for the threatened desert tortoise (*Gopherus agassizi*). The area is characterized by Mojave Desert climate and vegetation superimposed on colorful geological strata typical of the nearby Colorado

Plateau. Exposed sandstone formations have weathered and eroded to produce sandy soils punctuated by sandstone cliffs and outcrops (Mortensen and others 1977). Much of the area is public land administered by the Bureau of Land Management (BLM) where a variety of activities have taken place including livestock grazing, mining and off-road vehicle use (BLM, pers. comm.). Recently BLM management of the area has shifted pointedly toward habitat conservation in conjunction with (1) listing of the desert tortoise in this area as a threatened species on April 2, 1990 (USFWS 1990), (2) designating the area as critical habitat on February 8, 1994 (USFWS 1994) and (3) development of the Washington County, Utah Habitat Conservation Plan and the establishment of the 62,000-ac Red Cliffs Desert Reserve (BLM 2008; Smith and Rose 1996). Efforts are now underway to restore native vegetation on degraded lands resulting from earlier activities within the reserve (Rose and Duck 2001).

Restoration of native vegetation in the Mojave Desert ecosystem is complicated by a limited and fluctuating moisture supply for seed germination and establishment (Allen 1995; Bainbridge 2007; Stevens 2004). Cryptobiotic crusts, which provide soil stability, moisture retention and nutrient inputs to desert plants, are often deficient in disturbed sites where restoration is desired (Belnap 1995). Competition from non-native plants, especially invasive annuals such as brome grasses (*Bromus* spp.), is another factor that may limit restoration success in the Mojave Desert (Beatley 1966; Lovich and Bainbridge 1999). Although non-native annuals may contribute forage for the desert tortoise and other wildlife (Grover and DeFalco 1995; Schiffman 1994), their net effect on biodiversity and ecosystem functioning is typically negative because of their dominating effect and their production of fine fuel that facilitates the spread of damaging wildfires (Brooks and Berry 2006; Brooks and Matchett 2006; D'Antonio and Vitousek 1992).

Despite the challenge of the task, the BLM has carried out a series of restoration projects at the Red Cliffs Desert Reserve using reclamation technology (Bainbridge 2007; Thornburg 1982). An early project was initiated in 1992 with the goal of reclaiming and restoring two abandoned quarries. This paper reports the effects of this restoration project on vegetation establishment and subsequent vegetation development and dynamics over a period of 14 years following treatment, offering a longer-term view than a previous report spanning two years (McArthur and Sanderson 1996). Permanent plots provide the opportunity to assess vegetation changes and relate them to conditions during the monitoring period.

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Jeffrey E. Ott, Botanist, USDA, Forest Service, Rocky Mountain Research Station, Shrub Sciences Laboratory, Provo, UT

²E. Durant McArthur, Program Manager, USDA, Forest Service, Grasslands Shrublands and Desert Ecosystem Science Program, Shrub Sciences Laboratory, Provo, UT

³Stewart C. Sanderson, Research Geneticist, USDA, Forest Service, Rocky Mountain Research Station, Shrub Sciences Laboratory, Provo, UT.

MATERIALS AND METHODS

Study Area

The Twist Hollow restoration area (113° 36' 10" W longitude, 37° 09' 13" N latitude) is located on the east side of Utah State Highway 18 three km north of St. George, Washington County, Utah at an elevation of about 1010 m. The average annual precipitation at St. George during the 30-year period preceding this study was 204 mm with 64 percent of that falling Oct. 1 to March 31 (NOAA 2008). In common with other Mojave Desert locations precipitation patterns are erratic but the July-Sept. period receives approximately twice as much precipitation as the Apr.-June period (NOAA 2008). The area is topographically diverse with Navajo Sandstone outcrops towering over washes and wind-blown deposits of loamy fine sand of the Pintura Series (McArthur and Sanderson 1994; Mortensen and others 1977).

The primary treated site for the Twist Hollow restoration project, the Sand Pit, covers ca. three ac of sand deposits stretching from sandstone cliffs to the east and north to Highway 18 on the west (figure 1). Sand was mined from this area prior to the 1990s and its accessibility also led to widespread refuse dumping, off-road vehicle use, and target shooting activities. A secondary treatment site, the Rock Quarry, is located ca. 200 m northeast of the Sand Pit in a half-ac hollow of a northeast-facing sandstone outcrop above the Twist Hollow arroyo (figure 1). Soil and vegetation in the Rock Quarry hollow were disturbed during a period of rock quarrying prior to its closure at the time of the restoration project (BLM, pers. comm.).

Treatment

Physical site preparation and seeding of the Sand Pit and Rock Quarry sites were performed by BLM field crews from the Cedar City District Office. Prior to seeding which took place on December 7-8, 1992, the sites were cleared of refuse, sculptured with a bladed tractor and, in the case of the Rock Quarry, explosives, to give the area a natural appearance. The sites were disked before seeding (BLM, pers. comm.).

The sites were seeded with native plants with a rangeland drill at the rate of 19.5 lbs of seed per ac. The seed mix contained 8.0 lbs/ac Indian ricegrass (*Stipa hymenoides*), 2.5 lbs/ac sand dropseed (*Sporobolus cryptandrus*), 1.0 lbs/ac Palmer penstemon (*Penstemon palmeri*) and 2.0 lbs/ac each of galleta (*Hilaria jamesii*), gooseberryleaf globemallow (*Sphaeralcea grossulariifolia*), winterfat (*Krascheninnikovia lanata*) and fourwing saltbush (*Atriplex canescens*) (BLM, pers. comm.).

Data Collection

On November 25, 1992, prior to seeding, representative study plot locations were identified, referenced in relation to nearby topographic features and marked with buried steel rebars. Following seeding we were able to relocate the plots with the aid of a metal detector and the rebars were then positioned above ground for ease of relocation. Two of the Sand Pit plots were located in intact vegetation that was not disturbed by sculpting or seeding and served as control plots (one of these was added following seeding in 1993). One of the original Rock Quarry plots became partially covered by rubble and was subsequently omitted from our analysis. Here we report the results from 12 plots in 3 treatments: 8 Sand Pit Seeding plots, 2 Sand Pit Control plots and 2 Rock Quarry Seeding plots (figure 1).

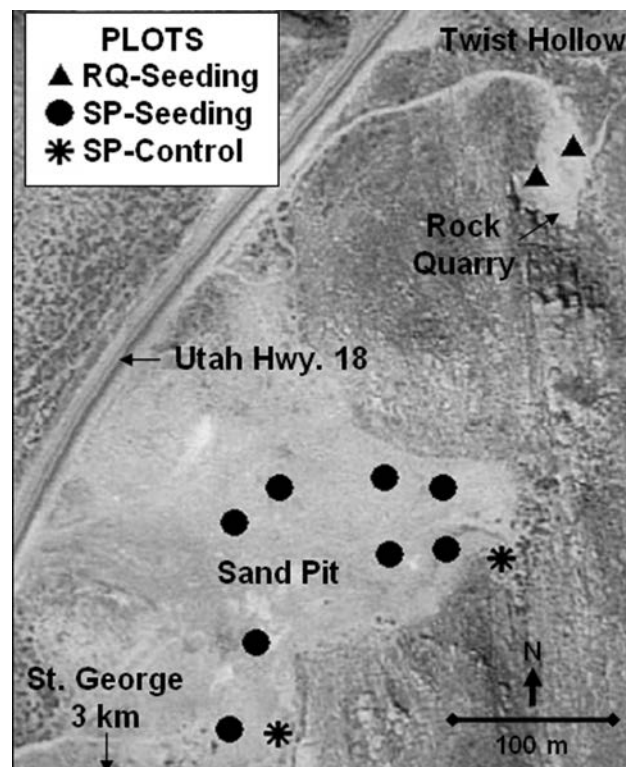


Figure 1—Aerial photograph (mid-1990s) of the Twist Hollow restoration site showing locations of the Sand Pit, Rock Quarry, and plots in each treatment. RQ=Rock Quarry, SP=Sand Pit. Source: Utah GIS Portal, <http://gis.utah.gov>.

Data were collected at the scale of a circular 100 m² plot and from eight systematically-located 1 m² subplots (figure 2). Vascular plant species composition was collected at each scale following the nomenclature of Welsh and others (1987), subsequently updated to Welsh and others (2003). At the 100 m² scale each species was assigned to one of the following percent cover classes: <1, 1 to 5, 6 to 25, 26 to 50, 51 to 75, 76 to 95, >95 (after Braun-Blanquet 1932). At the 8 m² scale direct estimates of percent cover were taken along with density counts for each species. Pre-treatment

baseline data were collected on November 25 to 28, 1992, followed by repeated episodes of post-treatment spring-season data collection on April 8 to 14, 1993; May 2 to 3, 1994; April 11 to 12, 1995; April 15 to 16, 1997; April 21 to 23, 1998; April 22 to 23, 2002; April 26 to 28, 2005; and May 7 to 8, 2007.

Data Analysis

For each species detected during the course of the study we calculated tabular summary statistics (frequency and cover in 100 m² plots; cover and density in 8 m² plots) by treatment and year. Geometric means of cover classes were used to obtain percent cover values for the 100 m² plots.

The localized nature of the study and small sample sizes limited our use of formal statistical analysis; however the Sand Pit Seeding treatment had a sufficiently large sample size ($n = 8$ plots) to test the statistical significance of observed changes in cover and density between successive data collection dates using a binomial sign test (Crawley 2007). For each plot and species (or group), changes between successive dates were classified as an increase, decrease, or no change. Omitting plots where a species was absent on both the beginning and ending date, the ratio of increases or decreases relative to the total number of plots was used as a test statistic. The following ratios lying within the realm of the data are significant at $\alpha=0.05$ according to a one-sided binomial sign test: 5:5, 6:6, 7:7, 7:8, and 8:8 (extracted from the `binom.test` function in R vers. 2.7.0; R Development Core Team 2008).

Additional insights into data patterns and environmental correlations were gained from exploratory analyses including hierarchical clustering and regression tree analysis (McCune and Grace 2002). These insights are described below although the results of these analyses are not explicitly presented.

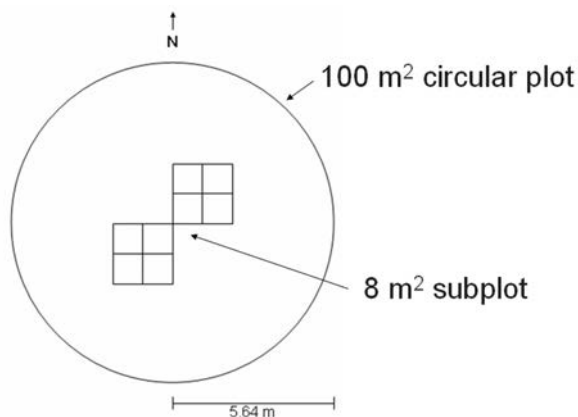


Figure 2—Vegetation Plot Layout used at Twist Hollow restoration site.

RESULTS

Different aspects of the Twist Hollow site vegetation and its variation over time are highlighted in tables 1 to 3. Table 1 lists each of the 105 vascular plant species recorded during the course of the study and its frequency (in 100 m² plots) by treatment and year. Mean cover values of species groups and selected species (seeded species and species with greater than 1 percent mean cover in any treatment/year/scale) are shown in table 2 for both 100 m² and 8 m² plot scales. Cover values mentioned in the text below are averages between the two scales unless otherwise indicated. Density estimates derived from the 8 m² plots are provided in table 3 for selected species (those recorded at least seven times in 8 m² plots of the seeded treatments over the course of the study). Figure 3 sets our data collection dates in the context of precipitation patterns over time at the nearby St. George weather station.

Pre-treatment and Intact Vegetation

Prior to treatment in the fall of 1992 the disturbed portions of the Sand Pit and Rock Quarry had sparse but variable vegetation cover dominated by annuals (tables 1 to 3). Total cover in the 100 m² plots ranged from zero to 35 percent with annuals contributing ca. 70 percent of the total (table 2). The dominant annual species were the exotics cheatgrass (*Bromus tectorum*) and Russian thistle (*Salsola tragus*) and the natives bur ragweed (*Ambrosia acanthicarpa*), gray sandplant (*Dicoria canescens*) little twistflower (*Streptanthella longirostris*) and whitestem blazingstar (*Mentzelia albicaulis*). Russian thistle cover was especially high in the Rock Quarry, exceeding 5 percent in both plots. Some plots of the Sand Pit has 1-5 percent cover of native perennial species (table 2) including Indian ricegrass, sand dropseed, longbracted trefoil (*Lotus plebius*), hairy goldenaster (*Chrysopsis villosa*) and broom snakeweed (*Gutierrezia sarothrae*). Indian ricegrass, although variable in cover, had the highest frequency of any perennial in the disturbed areas.

Relative to vegetation in the disturbed areas, nearby intact vegetation of control plots was characterized by a well-developed shrub layer and a denser herbaceous layer concentrated under or near shrub canopies (tables 1 to 3). The 1992 Sand Pit Control plot, representing a shallow sand environment, had ca. 23 percent shrub cover dominated by sandsage (*Artemisia filifolia*) with lesser amounts of blackbrush (*Coleogyne ramosissima*), range ratany (*Krameria parvifolia*) Nevada ephedra (*Ephedra nevadensis*), burrobrush (*Ambrosia salsola*), bush encelia (*Encelia frutescens*) and broom snakeweed (tables 1 to 2). The herbaceous layer was dominated by cheatgrass with ca. 16 percent cover (table 2) and density of ca. 664 m⁻², many times higher than recorded in any disturbed plot (table 3). Indian ricegrass and Underwood spikemoss (*Selaginella underwoodii*) were the dominant herbaceous perennials

with ca. 3 percent cover each, and sand dropseed was also present at ca. 1 percent cover (tables 1 to 2). The additional Sand Pit Control plot, added in 1993, captured a deeper sand environment that was also dominated by sandsage and cheatgrass. This plot contained fewer shrub species and the dominant perennial herbaceous species were Indian ricegrass, big galleta (*Hilaria rigida*), longbracted trefoil and Mojave croton (*Croton californicus*); and to a lesser extent sand verbena (*Abronia fragrans*), hyalineherb (*Hymenopappus filifolius*) hairy goldenaster and sand dropseed (tables 1 to 3).

General Treatment Effects

The sculpting and disking treatment had the effect of removing most of the existing vegetation from the Sand Pit and Rock Quarry although residual mature plants were detected in a few plots of the Sand Pit where shallow rocky soils limited the effectiveness of the treatment. Through residual re-growth and new seedlings most of the pre-treatment species had re-established by the spring of 1993. A few species that had been present in trace amounts in 1992, such as blackbrush, bush muhly (*Muhlenbergia porteri*), perennial milkvetch (*Astragalus* spp.) and rattlesnake-weed (*Chamaesyce albomarginata*), did not reappear following treatment (table 1). Other species, mostly annuals, were first detected in 1993, including the exotic species red brome (*Bromus rubens*) and Mediterranean grass (*Schismus barbatus*) in the Sand Pit (tables 1 to 3). Seedling establishment of both annuals and perennials was likely enhanced by above-average precipitation during the winter of 1992 to 93 and additional periods of high precipitation lasting through 1998 (figure 3).

A ca. 5 percent reduction in mean vegetation cover following treatment between fall 1992 and spring 1993 belies cover decreases in some plots and increases in others resulting in less variance in cover within the treated area. Vegetation cover rebounded in subsequent years and exceeded pre-treatment levels by 1995 in the Sand Pit and 1997 in the Rock Quarry (table 2). These increases cannot be attributed entirely to treatment effects because cover increases were also observed in the control plots during this period (table 2).

Much of the new cover following treatment consisted of annuals, particularly cheatgrass. Cheatgrass densities declined in the Sand Pit following treatment but rose from ca. 8 to 87 m⁻² between 1993 and 1995. By 1995 cheatgrass cover had reached ca. 9 percent in the sand pit and 5 percent in the Rock Quarry, then remained stable in the Rock Quarry through 1998 while declining to ca. 3 percent in the Sand Pit. In general, greater increases of cheatgrass cover and density were observed in plots with greater rock cover. Despite its expansion following treatment, mean cheatgrass cover and density in the treated plots remained lower than the control plots throughout the study (tables 2 to 3). Red brome remained subordinate to cheatgrass,

reaching a maximum of 1 percent cover in the Rock Quarry in 1995 (table 2). Russian thistle cover in the Rock Quarry fell to ca. 1 percent following treatment and remained at or below this level in subsequent years, although this may be an artifact of the underdeveloped stage of this species in springtime when sampling took place. High densities of Russian thistle (for example, 17 m⁻² in the Rock Quarry in 1993; table 3) suggest that this species likely remained dominant later in the season.

Seeded Species Establishment

Establishment success of seeded plants varied between species. Of the seeded grasses, Indian ricegrass had the greatest response following treatment, regaining pre-treatment levels of mean cover (in 100 m² plots) and density by the spring of 1993. By 1995, Indian ricegrass cover in the Sand Pit had risen to 2 percent and density to 1.1 m⁻²; and by 1997 in the Rock Quarry, 1 percent and 2.2 m⁻² (tables 2 to 3). Residual plants and/or seeds may be partially responsible for Indian ricegrass establishment, especially in the Sand Pit. Sand dropseed also appears to have persisted in some Sand Pit plots due to residual re-establishment following treatment, although it was lost from other plots between 1992 and 1993 (table 1). A flush of new sand dropseed seedlings then appeared in both the Sand Pit and Rock Quarry in 1994 (table 3), presumably originating largely from the seeding. However, sand dropseed cover remained below one percent in all treated plots and its frequency and density declined after 1994 (tables 1 to 3). Galleta initially became established in only one plot of the Rock Quarry where it persisted with less than 1 percent cover for several years. New galleta seedlings were later detected in three Sand Pit plots in 2005 (tables 1 to 2) but whether these are derived from the seeding is unknown.

Seeded fourwing saltbush became established in both the Sand Pit and Rock Quarry by the spring of 1993 but subsequently followed different trajectories in these two areas. After emerging in six Sand Pit 100 m² plots with a density of 0.2 m⁻² in 1993, fourwing saltbush declined to four plots and 0 m⁻² in 1994 and remained insignificant in the Sand Pit for the remainder of the study (tables 1 to 3). In the Rock Quarry, on the other hand, frequency increased from 1 to 2 and density from 0.4 to 0.5 m⁻² between 1993 and 1994 (tables 2 to 3). Density then declined to ca. 0.15 m⁻² for the remainder of the 1990s in the Rock Quarry while cover of surviving individuals rose to ca. 1 to 2 percent (tables 2 to 3).

The seeded forb Palmer penstemon was most successful in the Rock Quarry where it attained a density of 2.7 m⁻² and approached 1 percent cover in 1997 (tables 2 to 3). Palmer penstemon also became established in seven of the Sand Quarry plots by 1995 but declined in frequency in the Sand Pit thereafter (table 1).

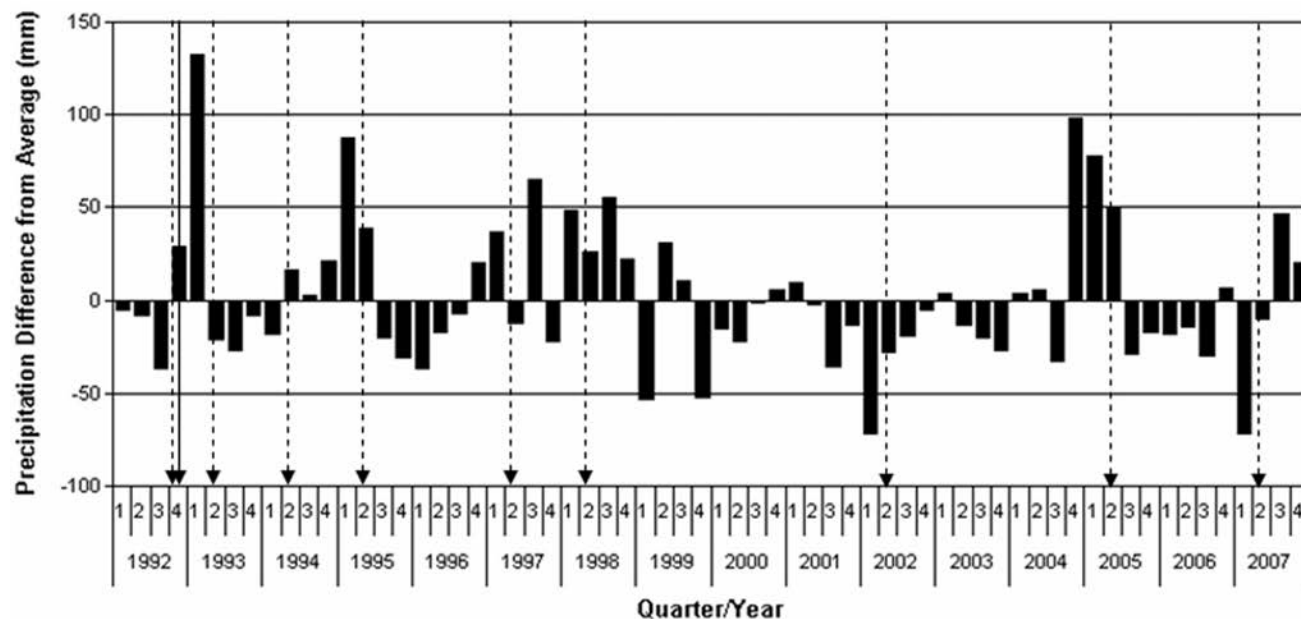


Figure 3—Precipitation patterns at the St. George weather station, Washington County, Utah, 1992–2007. Bars show total precipitation above or below average precipitation (30-year average, 1962 to 1992) calculated separately for each quarter. Dashed arrows are approximate dates of data collection at the Twist Hollow restoration site; solid arrow is treatment date. Source: NOAA (2008).

Emergence of seeded gooseberryleaf globemallow was delayed until 1994 in the Rock Quarry and 1995 in the Sand Pit. This species remained at low densities (less than 0.06 m^{-2}) with less than 1 percent cover before dropping out by the end of the study (tables 2 to 3), in contrast to the congener desert globemallow (*Sphaeralcea ambigua*) which, although not seeded, became established and increased in cover in some plots over the course of the study (tables 1 to 2). Desert globemallow was observed growing in nearby rock outcrops and likely recruited from these areas.

Winterfat was not detected in any plots following treatment and the reasons for its failure to establish are unknown. Other species appear to have seeded origin and may have been unintended components of the seed mix. The latter include the shrub rubber rabbitbrush (*Chrysothamnus nauseosus*), which gained prominence in the Rock Quarry, and the forbs alfalfa (*Medicago sativa*) and yellow sweetclover (*Melilotus officinalis*) which had an irregular and impermanent presence in both the Sand Pit and the Rock Quarry in the years following treatment (tables 1 to 3).

Multi-year Trends

As alluded to above, vegetation changes in the years following treatment can be related to precipitation patterns during this period (cf. Beatley 1974, 1980; Brooks and Berry 2006; Hereford and others 2006; Webb and others 1987). Seedling establishment and growth were favored by above-average precipitation during the winter immediately following seeding, and although the next winter was drier than normal, repeated periods of high precipitation (figure

3) led to increases in cover and density through 1998 (table 2). However, the strong El Niño event that brought high rainfall in 1998 was immediately followed by a strong La Niña event (WRCC 2008), initiating a period of drought that lasted through our next round of sampling in 2002 (figure 3). Cover and density of cheatgrass and other annuals declined dramatically between 1998 and 2002 in both treated and control plots, and many of the annuals that had been present in previous years were absent altogether (tables 1 to 3). In 2005 we captured an equally dramatic but opposite extreme of exceptionally high annual cover and density (tables 2 to 3) in consequence of another El Niño event (WRCC 2008), followed by a return to drought conditions in 2007 (figure 3). The post-2005 drought impacted cheatgrass less in the Rock Quarry, where cover rose to ca. 9 percent and density 222 m^{-2} 2007, than in the Sand Pit where it dropped to ca. 2 percent and 41 m^{-2} (tables 2 to 3). Some annuals of earlier importance (red brome, bur ragweed, Russian thistle, gray sandplant, alfalfa, yellow sweetclover) were virtually absent at the conclusion of the study in 2007 (tables 1 to 3).

Perennial grass cover also declined from 1998 to 2002 and did not rebound in 2005 or 2007 (table 2), despite slight increases in seedling density of Indian ricegrass in 2002 and sand dropseed in 2005 and 2007 (table 3). Other perennials varied in their responses during this period, with some declining such as rubber rabbitbrush and longbracted trefoil (*Lotus plebius*) and others increasing such as broom snakeweed (*Gutierrezia sarothrae*) and hairy goldenaster (*Chrysopsis villosa*) (tables 2 to 3). By 2007 the latter two species were co-dominant with fourwing saltbush at the

Rock Quarry, each having 1 to 2 percent cover (table 2). Broom snakeweed was the species with the highest mean cover (ca. 3 percent) in the Sand Pit Seeded treatment in 2007, and hairy goldenaster also became prominent in some of the rockier Sand Pit plots. In sandier plots, sandsage cover and density rose between 2002 and 2005, and cover remained high in 2007 despite a slight reduction in density (tables 2 to 3). In one deep-sand plot near the edge of the Sand Pit, sandsage attained ca. 10 percent cover by 2007, a value that approached sandsage cover values of the control plots.



Figure 4—Photographs of Rock Quarry (a) and Sand Pit (b) at the Twist Hollow restoration site in June 2008 after the conclusion of this study. Note large fourwing saltbush (*Atriplex canescens*) in treated area below quarry face in (a); abundance of broom snakeweed (*Gutierrezia sarothrae*) in (b). Fenceline shown in (b) restricts vehicular traffic from entering the site.

DISCUSSION

The objectives of the 1992 Twist Hollow treatment included aesthetic enhancement of the site, soil stabilization, wildlife (especially desert tortoise) habitat improvement, and native plant community restoration (BLM, pers. comm.). To varying degrees each of these objectives was realized over the course of 14 years following treatment although not necessarily in the manner anticipated. The role of the seeded plants in meeting these objectives differed depending on the time frame of observation. In our earlier assessment of the Twist Hollow treatment (McArthur and Sanderson 1996) we optimistically noted increasing trends in several seeded species on the treated substrate following periods of high precipitation. Because the early period of seed germination and plant establishment is typically considered the most sensitive and crucial period of growth (Stevens 2004), it was easy to conclude that established plants would persist over the long term. This conclusion proved partially correct inasmuch as some individuals of seeded species persisted through the duration of the study. However, with localized exceptions such as fourwing saltbush at the Rock Quarry, the seeded species were generally declining when viewed from the time frame of the latter years of the study when drought conditions prevailed.

In hindsight, the species selected for seeding might have been chosen differently, given the poor performance of some such as gooseberryleaf globemallow and galleta. Seed was obtained from commercially-available stock possibly derived from ecotypes poorly adapted to local conditions. However, the use of local seed sources may not have guaranteed greater seeded plant persistence. Local native populations were not immune to the effects of the harsh conditions of the post-1998 drought periods, as evidenced by declines in Indian ricegrass and other native species in the control plots (tables 1 to 3).

Those species that did establish and increase despite the drought, such as sandsage, broom snakeweed, hairy goldenaster and desert globemallow, might well be given greater attention in future restoration projects in the Red Cliffs environment. Colonization by these species was likely facilitated by the relatively small size of the quarrying disturbances and their proximity to intact vegetation. Assisting this colonization process through artificial seeding may be beneficial, especially where the disturbances are larger or local seed sources more limited. Sandsage is a good candidate for artificial seeding because it has been seeded elsewhere and its cultivation requirements are known (McArthur and Stevens 2004; McArthur and Taylor 2004); however, its natural range includes only the northeastern portion of the Mojave Desert (NRCS 2008) and it thus may not be suitable for many Mojave Desert sites.

In terms of site characteristics, species abundance and composition the intact vegetation we recorded at the Twist Hollow Sand Pit closely resembles sandsage communities of Washington County, Utah described by Rasmussen and Brotherson (1986). These authors noted that fourwing saltbush, blackbrush and creosote bush, though present in adjacent rocky, gravelly or clayey sites, were virtually absent from sandy sites where sandsage communities occurred (Rasmussen and Brotherson 1986). Of our two Sand Pit control plots, the deep sand plot was clearly aligned with these sandsage communities while the shallow sand plot had characteristics intermediate between sandsage and blackbrush communities. In the treated areas sandsage establishment was highest on deep sand of the Sand Pit, in contrast to fourwing saltbush on the rockier substrate of the Rock Quarry, suggesting different habitat preferences for each species related to substrate. Within sandsage communities, Rasmussen and Brotherson (1986) noted differences in species composition in the sandsage canopy zone relative to interspaces where Indian ricegrass was the dominant perennial. Sandsage and Indian ricegrass are apparently incompatible in close proximity but complement each other at the broader scale. Our results indicate that drought may play a role in determining the balance between these two species because sandsage appeared to be more drought-tolerant than Indian ricegrass. Elsewhere in its range, sandsage is commonly associated with drier and more overgrazed sites relative to grasses (Ramaley 1939; Sims and others 1976).

Perennial species are justifiably the focus of restoration projects in arid environments because of their stability and soil protection capacity. However, annuals are also important in terms of plant biodiversity in the Mojave Desert and are an important component of the desert tortoise diet (Grover and DeFalco 1995; Oftedal 2002). Of particular importance for desert tortoise nutrition are plant species rich in protein relative to potassium (Oftedal 2002, 2005). Many annuals have this property including several of the annual species and genera recorded following treatment at our study site such as desert globemallow, Pursh's plantain (*Plantago patagonica*), small-flowered milkvetch (*Astragalus nuttallianus*), storksbill (*Erodium cicutarium*), tansymustard (*Descurainia* spp.) and whitestem blazingstar (Oftedal 2002; Oftedal and Leuteritz 2006). The perennial forb longbracted trefoil, which was common at the Sand Pit before and after treatment during wet years, also likely has a high protein to potassium ratio by virtue of its nitrogen-fixing capacity. The exotic annual grasses (cheatgrass, red brome and Mediterranean grass), though readily eaten by desert tortoises, are nutritionally inferior because of their relatively low protein to potassium ratios (Oftedal 2002, 2005).

Cheatgrass re-establishment and dominance in the treated area was perhaps an unavoidable consequence of a residual seed bank and dense populations of this species in the

surrounding vegetation. Nevertheless, the consistently lower density and cover of cheatgrass in the treated plots relative to the control plots suggests limitations on cheatgrass imposed by the treated substrate. The control plots had not only denser populations providing more seed, but also more litter and cryptobiotic crust cover than the treated plots. Litter is known to enhance cheatgrass by providing safe sites for seed germination (Evans and Young 1984), while cryptobiotic crusts enhance growth through nutrient inputs (Belnap 1995; Pendleton and others 2003). Vesicular-arbuscular mycorrhizae are also likely to have been more abundant in the undisturbed soils, benefiting both cheatgrass and perennial species (Bainbridge 2007; Knapp 1996). Our observation that cheatgrass establishment was higher on rockier portions of the treated area, such as the Rock Quarry, suggests that rocky microtopography may have partially compensated for litter in providing safe sites for cheatgrass germination. Mycorrhizal establishment over time may have contributed to the decline in the non-mycotrophic species Russian thistle that paralleled the increase of cheatgrass at the Rock Quarry (Allen and Allen 1988).

Because the factors that favor establishment of desired Mojave Desert plant species generally also favor invasive annual grasses such as cheatgrass and red brome, successful restoration of native vegetation remains a complicated challenge. The beneficial effects of high rainfall years as a boon for native plant establishment and productivity can be outweighed by detrimental effects of explosive increases of these grasses. In 2005, for example, dense brome grass resulting from high precipitation fueled catastrophic fires throughout the Mojave Desert (Matchett 2006), including 14,741 ac that burned in the Red Cliffs Desert Reserve (McLuckie and others 2006). Innovative new strategies may be necessary to conserve the integrity of Mojave Desert vegetation from the threats of invasive species and wildfire (cf. BLM 2006).

Complete vegetation recovery on disturbed sites in arid regions is generally expected to take much longer than the 14-year timespan of this study (Bainbridge 2007). In light of the slowness of the process and the potential for setbacks, the Twist Hollow restoration project can be viewed as relatively successful. The seeded species successfully established and provided perennial cover in the early years following treatment while other species expanded into this role in the later years. Desert tortoises in areas adjacent to the treated areas currently appear to be thriving (BLM, pers. comm.). From an aesthetic standpoint the treated areas now blend reasonably well with their surroundings (figure 4). Fences constructed to restrict vehicular access (figure 4b) appear to have played an important role in reducing human disturbance and allowing recovery to take place. The site now contributes to the image of the Red Cliffs reserve as a scenic natural area, and with proper management may continue to do so.

ACKNOWLEDGEMENTS

We thank the U.S. Department of the Interior, Bureau of Land Management for financial and physical assistance. We

appreciate the help of Scott Belfit, Todd Esque, Gary Jorgensen, Clyde Blauer, Lance Stavast, Mark Burnham and Bob Douglas in various ways.

Table 1—Frequency (number of occurrences in 100 m² plots) by treatment and data collection year for species recorded at the Twist Hollow restoration site.

	Rock Quarry Seeded (2 plots)									Sand Pit Seeded (8 plots)									Sand Pit Control (2 plots)								
	2 ^d	3	4	5	7	8	2	5	7	2	3	4	5	7	8	2	5	7	2	3	4	5	7	8	2	5	7
-----Shrubs-----																											
<i>Ambrosia salsola</i>																			1	2	2	2	2	1			1
<i>Artemisia filifolia</i>										1	1	2	2	2	2	2	3	2	1	2	2	2	2	2	2	2	2
<i>Atriplex canescens</i> (s) ^a		1	2	2	2	2	2	2	2		6	4	3	3	3	3	3	3									
<i>Bassia prostrata</i> ^b (s?)				1	1	1																					
<i>Chrysothamnus nauseosus</i> (s?)			2	2	2	2	2	1																			
<i>Coleogyne ramosissima</i>						1			1	1						1	1	1	1	1	1	1	1	1	1	1	1
<i>Encelia frutescens</i>										2	1	1	1	1	1				1	1	1	1	1	1	1	1	1
<i>Ephedra nevadensis</i>																			1	2	2	2	2	2	2	2	2
<i>Ephedra viridis</i>				2	1	2	2	2															1		1	1	1
<i>Eriogonum fasciculatum</i>				1	1									1						2	1	1	1	1	1	1	1
<i>Fraxinus anomala</i>						1																					
<i>Gutierrezia sarothrae</i>			2	1	1	1	2	1	1	3	4	4	6	7	7	8	7	7	1	1	1	1	1	1	1	1	1
<i>Krameria erecta</i>																			1	1	1	1	1	1	1	1	1
<i>Opuntia erinacea</i>														1													
<i>Opuntia polyacantha</i>						1									1	1											
<i>Prunus fasciculata</i>						1				1		1	2	1	1	1	1	1		1	1	1	1	1	1	1	1
<i>Rhus aromatica</i>																						1					
-----Perennial Grasses-----																											
<i>Agropyron cristatum</i> ^b (s?)				1																							
<i>Aristida purpurea</i>									1		1	1		1		2	2	2	1								
<i>Bouteloua gracilis</i>													1														
<i>Elymus elymoides</i>													1														
<i>Elymus</i> spp. ^c (s?)			1	1	2									3													
<i>Hilaria jamesii</i> (s)			1	1	1	1	1									3	2										
<i>Hilaria rigida</i>																				1	1	1	1	1	1	1	1
<i>Muhlenbergia porteri</i>										1																	
<i>Sporobolus cryptandrus</i> (s)			1			1	2		2	4	2	7	5	5	6	5	1	5	1	2	2	2	2	2	2	2	2
<i>Stipa comata</i>													1														
<i>Stipa hymenoides</i> (s)	1	2	2	2	2	2	2	2	2	7	8	8	8	8	8	8	8	8	1	2	2	2	2	2	2	2	1
-----Perennial Forbs-----																											
<i>Abronia fragrans</i>										1	1	5	3	3	2	1	2	2		1	1	1	1	1	1	1	1
<i>Allium nevadense</i>								1			1	1		1	1								1	1		1	
<i>Androstephium breviflorum</i>																										1	
<i>Astragalus</i> spp.										2																	
<i>Baileya multiradiata</i>													1		1	3	6	5						1		1	1
<i>Calochortus flexuosus</i>																						1					
<i>Chamaesyce albomarginata</i>										2									1								

Table 1 (Continued)—Frequency by treatment and data collection year.

	Rock Quarry Seeded (2 plots)									Sand Pit Seeded (8 plots)									Sand Pit Control (2 plots)								
	2 ^d	3	4	5	7	8	2	5	7	2	3	4	5	7	8	2	5	7	2	3	4	5	7	8	2	5	7
<i>Chrysopsis villosa</i>	2	1	2	2	2	2	2	2	2	2	3	5	5	6	5	6	6	6		1	1	1	1	1	1		
<i>Convolvulus arvensis</i> ^b											1																
<i>Croton californicus</i>			1	1	1	1	1	1		3	4	5	4	4	4	4	4	4	1	1	2	1	1	1	1	1	1
<i>Cynanchum utahense</i>																		1			1		1	1		1	2
<i>Dichelostemma pulchellum</i>														1								1		1	1		
<i>Eriogonum inflatum</i>															1	1	3	3									
<i>Erioneuron pilosum</i>										1	1	1	2		1	1											
<i>Hymenopappus filifolius</i>																				1	1	1	1	1	1	1	1
<i>Linum perenne</i>												1	1														
<i>Lotus plebeius</i>						1		1		5	5	6	7	8	7	3	7	4	1	1	1	1	1	1		1	
<i>Mirabilis multiflora</i>											2																
<i>Oenothera pallida</i>													1	1	1	1											
<i>Penstemon eatonii</i>													1				2										
<i>Penstemon palmeri</i> (s)			1	1	1	1	1	1	1			5	7	6	5	1	4	1									
<i>Sanguisorba minor</i> ^b (s?)											1	1	1														
<i>Selaginella underwoodii</i>																			1								
<i>Senecio douglasii</i>							1	1	1																		
<i>Sphaeralcea ambigua</i>						1	1	1	1			6	3	2	1	3	3	1									
<i>Sphaeralcea grossulariifolia</i> (s)			1	1	2	2	1						4	5	5	3	2							1			
<i>Stephanomeria pauciflora</i>			1	1	1	1	1	1	1		1	2	3	5	5	5	5	5									
-----Annual Grasses-----																											
<i>Bromus diandrus</i> ^b								2				1	1				2					1				1	
<i>Bromus rubens</i> ^b			1	1	2		1	2			6	6	3	8	8	1	8				1	2	2		1	1	
<i>Bromus tectorum</i> ^b	2	2	2	2	2	2	2	2	2	6	8	8	8	8	8	7	8	8	1	2	2	2	2	2	2	2	2
<i>Festuca octoflora</i>					2	1		2			1	3	5	6	6	3	8			1	1	1	1	1		1	1
<i>Hordeum</i> spp. ^c											1	2	1	2	1												
<i>Schismus barbatus</i> ^b								1			2	5	7	7	7	2	8	8						1	2		2
-----Annual Forbs-----																											
<i>Ambrosia acanthicarpa</i>	1	2	2	2	2	2		1		6	8	8	8	8	8	5	8		1	1	1				1	1	
<i>Amsinckia intermedia</i>													1														
<i>Astragalus nuttalianus</i>			1	2	2	2		2						3	4		6	2			1						
<i>Bassia scoparia</i> ^b																											
<i>Chamaesyce</i> spp.														1													
<i>Chorisporea tenella</i> ^b					2						4			1									1				
<i>Cryptantha barbigera</i>																	2										
<i>Cryptantha circumsissa</i>												4			1					2	2		1	1			
<i>Cryptantha crassisepta</i>								1																			
<i>Cryptantha micrantha</i>											2		5	7	6	4	6	3		2	1	2	2	2	2	2	2
<i>Cryptantha pterocarya</i>			1		2			2			1	3	3	4	5		4			1	2	2	2	2		2	

Table 1 (Continued)—Frequency by treatment and data collection year.

	Rock Quarry Seeded (2 plots)									Sand Pit Seeded (8 plots)								Sand Pit Control (2 plots)									
	2 ^d	3	4	5	7	8	2	5	7	2	3	4	5	7	8	2	5	7	2	3	4	5	7	8	2	5	7
<i>Datura wrightii</i>	1		1	1		1	2	1																			
<i>Descurainia</i> spp. ^c		1		1	1	1		1			3	1	5	3			3			1	1	1					
<i>Dicoria canescens</i>										2	7	4	6	3	3	1	3			1	1		1				
<i>Dithyrea wislizenii</i>												1	1		1			1		2	1	1		2		1	2
<i>Draba verna</i>								1									2										
<i>Eriastrum</i> spp.								1				4	3	5	4		8	7	1	2	2	2	2	2	1	2	1
<i>Eriogonum</i> spp.		1						1						2	1	3	3										
<i>Eriophyllum lanosum</i>																											
<i>Eriophyllum wallacei</i>								1			1	1	2	3	1		6	1		2	1	2	2	2	2	2	1
<i>Erodium cicutarium</i> ^b	1	1	2	2	2	2		2	2	1	2	8	8	8	8	2	8	7	1	2	1	2	2	2		2	2
<i>Gilia</i> spp.		1	2	1	2	2		2	1				2	1	2		3						1	1		1	
<i>Lepidium densiflorum</i>													1				2				1						
<i>Linanthus bigelovii</i>													1	2	2	1					1	1	1	1			
<i>Linanthus demissus</i>																	8							1		2	
<i>Lupinus concinnus</i>										1	4	2	5	4	4		7	4		1	1	1	2			1	
<i>Malcolmia africana</i> ^b													1														
<i>Medicago sativa</i> ^b (s?)			1	1	1						1	1	2	1	1	1	1										
<i>Melilotus officinalis</i> ^b (s?)		1	1			1					1	1	1		1		1										
<i>Mentzelia albicaulis</i>	2	1	2	2	2	2	1	1	1									1									
<i>Microseris lindleyi</i>								1							1		3			1		1	2	2		1	
<i>Oenothera primiveris</i>			1							1									1								
<i>Palafoxia arida</i>												3	1								1						1
<i>Pectocarya setosa</i>																	4										
<i>Phacelia ambigua</i>													1														
<i>Plantago patagonica</i>																	4	1		1	1	1	1	1		1	1
<i>Polygonum aviculare</i>											1																
<i>Salsola tragus</i> ^b	2	2	2	2	2	2				7	7	8	7	8	5	1		1	1				1				
<i>Sisymbrium</i> spp. ^b		1	1	1		1		2		1	4	7	6	6	2		6		1					1			
<i>Streptanthella longirostris</i>										4	7	8	7	7	6		5	5	1	1	2	2	2	2		1	1
<i>Stylocline micropoides</i>																	3										
<i>Tragopogon dubius</i> ^b																										1	
<i>Tripterocalyx micranthus</i>					2																						

^a(s) indicates seeded species; (s?) were probably also introduced through the seed mix though not reported. ^bnon-native species. ^cnative and/or non-native species. ^dnumbers indicate last digit of data collection years in following sequence: 2=1992, 3=1993, 4=1994, 5=1995, 7=1997, 8=1998, 2=2002, 5=2005, 7=2007.

Table 2—Mean cover of selected vascular plant species and functional groups by treatment and data collection year at the Twist Hollow restoration site. Leading cover values are for 100 m² plots; values in parentheses are for 8 m² subplots. Values are rounded to the nearest digit; "+" indicates values less than 0.5 and blank values are zeros. An asterisk denotes values significantly higher or lower than the corresponding value of the previous data collection year according to a one-sided binomial sign test at alpha=0.05 (see text for details). Treatment RQ-Seeded=X, SP-Seeded=Y, and SP-Control=Z.

Treatment ^c		1992		1993		1994		1995		1997		1998		2002		2005		2007	
-----Shrubs-----																			
<i>Artemisia filifolia</i>	Y	+		+		+		+	(+)	+		+	(+)	+	(+)	1	(1)	1	(1)
	Z	11 ^d	(16) ^d	11	(10)	11	(9)	23	(6)	11	(5)	11	(8)	11	(6)	7	(8)	11	(9)
<i>Atriplex canescens</i> (s) ^a	X			+	(+)	+	(+)	+	(+)	2	(1)	2	(1)	2	(1)	1	(+)	2	
	Y			+*	(+)	+		+		+		+		+		+		+	
<i>Chrysothamnus nauseosus</i> (s?)	X					+	(+)	+	(+)	+	(2)	+	(2)	1	(5)	+			
	X					+	(+)	+	(+)	+	(+)	+	(1)	+	(3)	1	(1)	1	(2)
<i>Gutierrezia sarothrae</i>	Y	+	(2)	1	(+)	1	(1)	2	(1)	2	(1)	2	(1)	3	(2)	2	(2)	3	(3)
	Z	+ ^d		+		+		+		+		+		+		+		+	
Total Shrubs	X			+	(+)	+	(+)	+	(+)	2	(3)	2	(4)	3	(9)	2	(1)	3	(2)
	Y	+	(2)	1	(+)	1	(1)	2	(1)	2	(1)	2	(1)	3	(2)	3	(3)	4	(4)
	Z	18 ^d	(30) ^d	16	(15)	17	(19)	28	(13)	16	(11)	15	(14)	17	(10)	9	(13)	12	(9)
-----Perennial Grasses-----																			
<i>Hilaria jamesii</i> (s)	X					+	(+)	+	(+)	+		+	(+)	+	(+)				
	Y															+		+	
<i>Sporobolus cryptandrus</i> (s)	X					+	(+)					+		+				+	
	Y	+	(+)	+	(+)	+	(+)	+	(+)	+	(+)	+	(+)	+		+		+	(+)
	Z	+ ^d	(2) ^d	+	(1)	1	(1)	+	(+)	1	(+)	1	(2)	1	(+)	+	(+)	+	(+)
<i>Stipa hymenoides</i> (s)	X	+		+	(+)	+	(+)	+	(1)	1	(1)	1	(2)	1	(1)	1	(1)	1	(+)
	Y	1	(2)	1	(+)	1	(1)*	2	(2)	2	(1)	2	(1)	1	(1)	+	(+)	+	(+)
	Z	2 ^d	(4) ^d	2	(3)	2	(6)	2	(4)	2	(1)	2	(2)	+	(+)	+	(+)	+	
Total Perennial Grasses	X	+		+	(+)	+	(+)	+	(1)	1	(1)	1	(2)	1	(1)	1	(1)	1	(+)
	Y	1	(2)	1	(+)	1	(1)*	2	(2)	2	(1)	2	(1)	1	(1)	+	(+)	+	(+)
	Z	2 ^d	(6) ^d	3	(4)	4	(7)	2	(4)	3	(1)	3	(4)	1	(+)	+	(+)	+	(+)
-----Perennial Forbs-----																			
<i>Chrysopsis villosa</i>	X	+		+	(+)	+	(+)	+	(1)	1	(2)	1	(2)	1	(1)	2	(1)	2	(1)
	Y	+	(+)	+	(+)	+	(+)	1	(+)	1	(+)	+	(+)	1	(1)	1	(1)	1	(1)
	Z	0 ^d	(0) ^d	+	(+)	+	(1)	+	(+)	+	(+)	+	(+)	+					
<i>Lotus plebeius</i>	X											+				+	(+)		
	Y	2	(1)	2	(1)	1	(1)	1	(1)	+	(1)	1	(2)	+	(+)*	+	(+)	+	(+)
	Z	+ ^d	(+) ^d	1	(+)	+		1	(+)	+	(+)	1	(+)			+	(+)		
<i>Penstemon palmeri</i> (s)	X					+	(+)	+	(+)	+	(1)	+	(+)	+	(+)	+	(+)	+	
	Y					+	(+)	+	(+)	+	(+)	+	(+)	+	(+)	+	(+)	+	(+)
<i>Sphaeralcea ambigua</i>	X											+	(+)	+	(+)	1	(+)	1	
	Y					+	(+)*	+	(+)	+	(+)	+		+	(+)	+	(+)	+	(+)
<i>Sphaeralcea grossulariifolia</i> (s)	X					+		+		+		+		+					
	Y							+	(+)	+		+	(+)	+	(+)	+			
	Z									+		+							
Total Perennial Forbs	X	+		+	(+)	+	(+)	+	(1)	1	(4)	1	(2)	1	(1)	3	(1)	4	(1)
	Y	2	(1)	2	(1)	1*	(1)	2	(1)	2	(1)	1	(2)	1	(1)	1	(2)	2	(1)*
	Z	2 ^d	(5) ^d	2	(1)	2	(3)	2	(2)	+	(1)	1	(1)	+	(+)	+	(+)	+	(+)
-----Annual Grasses-----																			
<i>Bromus rubens</i> ^b	X					+		1		1	(1)			+	(+)	+	(+)		
	Y			+	(+)	+	(+)	+	(+)	+	(+)	+	(+)	+		+	(1)*		(0)*
	Z	0 ^d	(0) ^d			+		+		+	(+)			+		+			
<i>Bromus tectorum</i> ^b	X	1	(+)	1	(1)	7	(2)	7	(2)	6	(2)	7	(2)	+	(+)	2	(17)	7	(11)
	Y	2	(1)	2	(1)	6	(3)*	12	(5)	9	(4)	5	(2)	+	(+)*	3	(5)*	2	(2)*
	Z	11 ^d	(21) ^d	11	(24)	11	(26)	48	(29)	48	(29)	23	(22)	+	(3)	2	(47)	2	(28)

Table 2 (Continued)—Mean cover of selected vascular plant species and functional groups by treatment and data collection year.

Treatment ^c		1992	1993	1994	1995	1997	1998	2002	2005	2007
Total Annual Grasses	X	1 (+)	1 (1)	7 (2)	8 (2)	7 (3)	7 (2)	+	(+)	2 (17)
	Y	2 (1)	2 (1)	6 (3)*	12 (5)	9 (4)	5 (2)	+	(+)*	3 (8)*
	Z	11 ^d (21) ^d	11 (25)	11 (26)	48 (29)	48 (29)	23 (22)	+	(3)	2 (47)
										3 (28)
Annual Forbs										
<i>Ambrosia acanthicarpa</i>	X	1 (+)	1	1 (1)	1 (1)	+	(1)	+	(1)	(0)*
	Y	3 (2)	3 (1)	+	(1)	+	(1)	+	(1)	+
	Z	+ ^d (+) ^d	+	+	(+)			+	(+)	+
<i>Dicoria canescens</i>	Y	3 (1)	1 (+)	+	(+)	+	(+)	+	(+)	+
	Z	0 ^d (0) ^d	+	+	(+)	+	(+)			
<i>Eriophyllum wallacei</i>	X									+
	Y		+	+		+		+	(+)	+
	Z	0 ^d (0) ^d	1 (1)	+	(+)	+	(2)	1 (5)	1 (2)	+
<i>Erodium cicutarium</i> ^b	X	+	(+)	+	(+)	+	(1)	+	(1)	+
	Y	+	(+)	+	(+)	+	(+)	+	(+)	+
	Z	+ ^d (1) ^d	+	+	(+)	+	(1)	1 (1)	1 (2)	+
<i>Medicago sativa</i> ^b (s?)	X			+		+	(+)	+	(+)	+
	Y		+	+		+		+	(+)	+
<i>Melilotus officinalis</i> ^b (s?)	X		+	+	(+)			+		
	Y		+	1 (1)	+	(1)	+	(1)	+	(+)
<i>Mentzelia albicaulis</i>	X	1	+	1 (1)	1 (1)	1 (1)	1 (1)	1 (1)	+	(+)
	Y									+
<i>Salsola tragus</i> ^b	X	11 (1)	1 (1)	+	(1)	+	(1)	+	(+)	
	Y	1 (1)	1 (+)	+	(+)	+	(+)	+	(+)	+
	Z	+ ^d				+	(+)			
<i>Sisymbrium spp.</i> ^b	X		+	(+)	+			+		+
	Y	+	+	(+)	+	(+)	+	(+)	+	(+)*
	Z	+ ^d (2) ^d						+		0*
<i>Streptanthella longirostris</i>	Y	1 (+)	+	(+)	+	(+)	+	(+)	0*	(0)*
	Z	2 ^d (4) ^d	+	(+)	+	(+)	+	(+)	+	(+)
Total Annual Forbs	X	13 (1)	2 (1)	2 (4)	3 (3)	2 (6)	1 (3)	1 (+)	+	(1)
	Y	6 (4)	4 (1)	2 (2)	1 (3)	+	(1)	+	(+)*	1 (2)*
	Z	2 ^d (8) ^d	2 (1)	1 (1)	1 (9)	6 (12)	3 (6)	+	(1)	1 (2)
Total Cover (multi-canopy)	X	15 (1)	3 (3)	9 (6)	11 (7)	14 (17)	12 (13)	7 (11)	9 (21)	18 (15)
	Y	12 (10)	10 (3)	11 (8)*	19*	(12)	16 (8)	11 (9)	4*	(4)*
	Z	36 ^d (70) ^d	35 (47)	36 (56)	82 (57)	73 (54)	46 (47)	18 (14)	3 (62)	36 (37)

^a(s) indicates seeded species; (s?) were probably also introduced through the seed mix though not reported. ^bnon-native species. ^ctreatment abbreviations: RQ-Seeded = Rock Quarry Seeded; SP-Seeded = Sand Pit Seeded; SP-Control = Sand Pit Control. ^dvalues shown for 1992 SP-Control are from a single plot, unlike other years that are the average of two SP-Control plots.

Table 3—Mean density per 100 m² (extrapolated from density counts in 8 m² plots) of selected vascular plant species by treatment and data collection year at the Twist Hollow restoration site. An asterisk denotes values significantly higher or lower than the corresponding value of the previous data collection year according to a one-sided binomial sign test at alpha=0.05 (see text for details).

	Treatment ^d	1992	1993	1994	1995	1997	1998	2002	2005	2007
Shrubs										
<i>Artemisia filifolia</i>	SP-Seeded	0	0	0	5	0	3	2	6	2
	SP-Control	100 ^c	137	494	106	119	81	44	19	31
<i>Atriplex canescens</i> (s) ^a	RQ-Seeded	0	38	50	19	13	19	13	6	0
	SP-Seeded	0	20*	0*	0	0	0	0	0	0
<i>Chrysothamnus nauseosus</i> (s?)	RQ-Seeded	0	0	6	6	6	6	6	0	0
<i>Ephedra viridis</i>	RQ-Seeded	0	0	0	0	0	0	6	6	19

Table 3 (Continued) —Mean density of selected vascular plant species by treatment and data collection year.										
	Treatment^d	1992	1993	1994	1995	1997	1998	2002	2005	2007
<i>Gutierrezia sarothrae</i>	RQ-Seeded	0	0	13	0	0	0	19	181	56
	SP-Seeded	47	89	77	172	203	141	144	880*	294
<i>Opuntia polyacantha</i>	RQ-Seeded	0	0	0	0	0	6	0	0	0
	SP-Seeded	0	0	0	0	0	2	2	0	0
-----Perennial Grasses-----										
<i>Aristida purpurea</i>	SP-Seeded	0	0	0	0	0	0	2	2	3
<i>Hilaria jamesii</i> (s)	RQ-Seeded	0	0	13	19	0	6	13	0	0
<i>Sporobolus cryptandrus</i> (s)	RQ-Seeded	0	0	19	0	0	0	0	0	0
	SP-Seeded	3	2	52	8	5	6	0	0	23
	SP-Control	26 ^c	25	44	31	13	1169	38	81	56
<i>Stipa hymenoides</i> (s)	RQ-Seeded	0	56	50	75	106	94	44	50	19
	SP-Seeded	59	78	156	113	222*	116	134	84	19
	SP-Control	62 ^c	94	156	119	100	19	13	6	0
-----Perennial Forbs-----										
<i>Abronia fragrans</i>	SP-Seeded	0	0	2	0	3	0	0	3	5
	SP-Control	0 ^c	306	319	150	69	113	31	44	6
<i>Allium nevadense</i>	SP-Seeded	0	2	0	0	5	6	0	0	0
	SP-Control	0 ^c	0	0	0	363	300	0	6	0
<i>Baileya multiradiata</i>	SP-Seeded	0	0	0	0	0	0	6	13	8
	SP-Control	0 ^c	0	0	0	0	0	0	13	6
<i>Chrysopsis villosa</i>	RQ-Seeded	0	13	13	56	244	513	38	563	275
	SP-Seeded	5	16	16	248	50	95	67	1081*	108
	SP-Control	0 ^c	113	31	44	63	6	0	0	0
<i>Croton californicus</i>	RQ-Seeded	0	0	6	6	6	6	13	6	0
	SP-Seeded	2	3	3	6	19	28	13	73	23
	SP-Control	0 ^c	38	25	44	6	25	0	31	6
<i>Eriogonum inflatum</i>	SP-Seeded	0	0	0	0	0	0	0	3	6
<i>Lotus plebeius</i>	RQ-Seeded	0	0	0	0	0	0	0	6	0
	SP-Seeded	41	173	105	794*	153	613	2*	180*	3*
	SP-Control	26 ^c	6	0	6	31	44	0	6	0
<i>Penstemon eatonii</i>	SP-Seeded	0	0	0	2	0	0	0	3	0
<i>Penstemon palmeri</i> (s)	RQ-Seeded	0	0	13	6	275	194	6	69	0
	SP-Seeded	0	0	3	6	2	2	2	13	2
<i>Sphaeralcea ambigua</i>	RQ-Seeded	0	0	0	0	0	6	6	0	0
	SP-Seeded	0	0	17*	3	3	0	2	13	2
<i>Sphaeralcea grossulariifolia</i> (s)	SP-Seeded	0	0	0	2	0	6	2	0	0
<i>Stephanomeria pauciflora</i>	RQ-Seeded	0	0	0	13	94	44	0	0	0
	SP-Seeded	0	0	2	5	22	9	25*	78	27
-----Annual Grasses-----										
<i>Bromus diandrus</i> ^b	RQ-Seeded	0	0	0	0	0	0	0	44	0
	SP-Seeded	0	0	0	2	0	0	0	6	0
	SP-Control	0 ^c	0	0	0	0	0	0	31	0
<i>Bromus rubens</i> ^b	RQ-Seeded	0	0	0	0	519	0	25	300	0
	SP-Seeded	0	95	16	2	41*	22	0	750*	0*
	SP-Control	0 ^c	0	0	0	19	0	0	0	0
<i>Bromus tectorum</i> ^b	RQ-Seeded	256	413	1481	3519	3869	4169	6	38438	22219
	SP-Seeded	1159	839*	2108*	8720*	2641*	8105*	513*	8022*	4098*
	SP-Control	66375 ^c	39663	182719	33125	29406	30125	3850	77563	32938
<i>Festuca octoflora</i>	RQ-Seeded	0	0	0	0	0	0	0	38	0
	SP-Seeded	0	11	2	22	77	161*	17*	3919*	0*
	SP-Control		613	781	813	469	375	0	2000	69
<i>Schismus barbatus</i> ^b	SP-Seeded	0	13	19	103*	147	150	2*	5167*	2983
	SP-Control	0 ^c	0	0	0	19	6	0	44	100

Table 3 (Continued)—Mean density of selected vascular plant species by treatment and data collection year.

	Treatment ^d	1992	1993	1994	1995	1997	1998	2002	2005	2007
-----Annual Forbs-----										
<i>Ambrosia acanthicarpa</i>	RQ-Seeded	6	0	3219	2013	1669	475	0	119	0
	SP-Seeded	73	1316*	2914	2161	1781*	1978	134*	161	0*
	SP-Control	12 ^e	6	6	0	0	0	31	38	0
<i>Astragalus nuttalianus</i>	RQ-Seeded	0	0	6	0	6	31	0	31	0
	SP-Seeded	0	0	0	0	2	9	0	17*	5
<i>Cryptantha micrantha</i>	SP-Seeded	0	3	0	277*	173	4597	902	1116	234*
	SP-Control	0 ^e	119	6	7031	12463	11281	3006	1294	250
<i>Cryptantha pterocarya</i>	RQ-Seeded	0	0	0	0	0	0	0	13	0
	SP-Seeded	0	0	2	11	14	22	0	11	0
	SP-Control	0 ^e	31	25	106	431	163	0	31	0
<i>Descurainia spp.^e</i>	RQ-Seeded	0	0	0	13	25	13	0	6	0
	SP-Seeded	0	2	0	33*	8	0	0	6	0
<i>Dicoria canescens</i>	SP-Seeded	117	263*	103*	120	50	42	2	13	0
	SP-Control	0 ^e	100	19	0	31	0	0	0	0
<i>Eriastrum spp.</i>	SP-Seeded	0	0	0	14	19	66	0	0	331*
	SP-Control	62 ^e	44	213	331	1275	244	0	0	31
<i>Eriogonum spp.</i>	SP-Seeded	0	0	0	0	0	6	5	113	0
<i>Eriophyllum wallacei</i>	SP-Seeded	0	0	0	0	0	2	0	16	0
	SP-Control	0 ^e	1300	644	3594	8563	8981	481	5838	81
<i>Erodium cicutarium^b</i>	RQ-Seeded	6	13	231	844	2519	2669	0	1625	625
	SP-Seeded	2	17	30	33	67	555*	9*	2302*	519
	SP-Control	237 ^e	88	38	169	506	775	0	1188	50
<i>Gilia spp.</i>	RQ-Seeded	0	0	163	19	88	419	0	456	6
	SP-Seeded	0	0	0	5	2	2	0	64	0
	SP-Control	0 ^e	0	0	0	6	6	0	6	0
<i>Linanthus bigelovii</i>	SP-Seeded	0	0	0	3	5	9	3	0	0
	SP-Control	0 ^e	0	6	31	6	6	0	0	0
<i>Linanthus demissus</i>	SP-Seeded	0	0	0	0	0	0	0	2386*	0*
	SP-Control	0 ^e	0	0	0	0	6	0	4500	0
<i>Lupinus concinnus</i>	SP-Seeded	2	2	0	0	5	2	0	89*	8
	SP-Control	0 ^e	0	0	0	6	0	0	0	0
<i>Medicago sativa^b (s?)</i>	RQ-Seeded	0	0	0	6	13	0	0	0	0
	SP-Seeded	0 ^e	14	0	0	0	2	0	2	0
<i>Melilotus officinalis^b (s?)</i>	SP-Seeded	0	0	8	1780	0	77	0	9	0
	RQ-Seeded	0	0	6	0	0	0	0	0	0
<i>Mentzelia albicaulis</i>	RQ-Seeded	0	0	344	513	238	213	6	13	0
	SP-Seeded	0	0	0	0	0	0	0	0	2
<i>Pectocarya setosa</i>	SP-Seeded	0	0	0	0	0	0	0	50	0
<i>Plantago patagonica</i>	SP-Seeded	0	0	0	0	0	0	0	17	16
	SP-Control	0 ^e	31	0	69	56	81	0	756	50
<i>Salsola tragus^b</i>	RQ-Seeded	69	1731	988	175	638	188	0	0	0
	SP-Seeded	27	505*	386	111	27	16	2	0	2
	SP-Control	0 ^e	0	0	0	6	0	0	0	0
<i>Sisymbrium spp.^b</i>	RQ-Seeded	0	6	13	0	0	0	0	81	0
	SP-Seeded	0	41	22	41	17	2	0	63*	0*
	SP-Control	100 ^e	0	0	0	0	0	0	0	0
<i>Streptanthella longirostris</i>	SP-Seeded	11	33	19	170	138	33	0*	61*	6*
	SP-Control	262 ^e	44	38	113	263	25	0	6	0

^a(s) indicates seeded species; (s?) were probably also introduced through the seed mix though not reported. ^bnon-native species. ^cnative and/or non-native species. ^dtreatment abbreviations: RQ-Seeded = Rock Quarry Seeded; SP-Seeded = Sand Pit Seeded; SP-Control = Sand Pit Control. ^evalues shown for 1992 SP-Control are from a single plot, unlike other years that are the average of two SP-Control plots.

REFERENCES

- Allen, E.B. 1995. Restoration ecology: limits and possibilities in arid and semiarid lands. In: Roundy, B.A., et al. comps. Proceedings: Wildland Shrub and Arid Land Restoration Symposium, 1993 October 19-21, Las Vegas, NV. Gen. Tech. Rep. INT-GTR-315. Ogden, UT: USDA, Forest Service, Intermountain Research Station: 7-15.
- Allen, E.B., Allen, M.F. 1988. Facilitation of succession by the nonmycorrhizal colonizer *Salsola kali* (Chenopodiaceae) on a harsh site: effects of mycorrhizal fungi. *American Journal of Botany* 75: 257-266.
- Bainbridge, D.A. 2007. A Guide for Desert and Dryland Restoration: New Hope for Arid Lands. Island Press, Washington DC. Beatley J.C. 1966. Ecological status of introduced brome grasses (*Bromus* spp) in desert vegetation of southern Nevada. *Ecology* 47: 548-554.
- Beatley, J.C. 1974. Phenological events and their environmental triggers in Mojave-desert ecosystems. *Ecology* 55: 856-863.
- Beatley, J.C. 1980. Fluctuations and stability in climax shrub and woodland vegetation of the Mojave, Great Basin and transition deserts of southern Nevada. *Israel Journal of Botany* 28: 149-168.
- Belnap J. 1995. Surface disturbances: Their role in accelerating desertification. *Environmental Monitoring Assessment* 37: 39-57.
- BLM (USDI, Bureau of Land Management). 2006. Vegetation Treatments EIS and Environmental Report. [Online]. Available: <http://www.blm.gov/nhp/spotlight/VegEIS> [July 9, 2008].
- BLM (USDI, Bureau of Land Management). 2008. Red Cliffs Desert Reserve. [Online]. Available: http://www.blm.gov/ut/st/en/fo/st_george/blm_special_areas/red_cliffs_desert.html [June 3, 2008].
- Braun-Blanquet, J. 1932. *Plant Sociology: the study of plant communities*. McGraw-Hill, New York. 439 p.
- Brooks, M.L., Berry, K.H. 2006. Dominance and environmental correlates of alien annual plants in the Mojave Desert, USA. *Journal of Arid Environments* 67: 100-124.
- Brooks, M.L., Matchett, J.R. 2006. Spatial and temporal patterns of wildfires in the Mojave Desert, 1980-2004. *Journal of Arid Environments* 67: 148-164.
- Crawley, M.J. 2007. *The R Book*. John Wiley and Sons Ltd, Imperial College London at Silwood Park, UK. 942 p.
- D'Antonio, C.M., Vitousek, P.M. 1992. Biological invasions by exotic grasses, the grass fire cycle, and global change. *Annual Review of Ecology and Systematics* 23: 63-87.
- Evans R.A., Young, J.A. 1984. Microsite requirements for downy brome (*Bromus tectorum*) infestation and control on sagebrush rangelands. *Weed Science* 32: 13-17.
- Grover, M.C., DeFalco, L.A. 1995. Desert Tortoise (*Gopherus agassizi*): Status-of-Knowledge Outline With References. INT-GTR-316. Ogden, UT: USDA, Forest Service, Intermountain Research Station. 134 p.
- Hereford, R., Webb, R.H., Longpre, C.I. 2006. Precipitation history and ecosystem response to multidecadal precipitation variability in the Mojave Desert region, 1893-2001. *Journal of Arid Environments* 67:13-34.
- Knapp, P.A. 1996. Cheatgrass (*Bromus tectorum* L) dominance in the Great Basin Desert—history, persistence, and influences to human activities. *Global Environmental Change—Human and Policy Dimensions* 6: 37-52.
- Lovich J.E., Bainbridge, D. 1999. Anthropogenic degradation of the southern California desert ecosystem and prospects for natural recovery and restoration. *Environmental Management* 24: 309-326.
- Matchett, J.R. 2006. Summary of the summer 2005 Mojave Desert fire season: when, where, and why fires occurred. Abstracts of the 2006 Desert Tortoise Council 31st Annual Meeting and Symposium, 2006 February 17-20, Tucson, AZ. Available: <http://www.deserttortoise.org/abstract/abstracts2006/2006abstracts.pdf> [June 3, 2008].
- McArthur, E.D., Sanderson, S.C. 1994. Nutritive Quality and Mineral Content of Potential Desert Tortoise Food Plants. Research Paper INT-473. Ogden, UT: USDA, Forest Service, Intermountain Research Station. 26 p.
- McArthur, E.D., Sanderson, S.C. 1996. Twist Hollow: a Mohave Desert restoration project. In: West, N.E., ed. *Rangelands in a sustainable biosphere: proceedings of the Fifth International Rangeland Congress*. Denver, CO: Society for Range Management: 350-351.
- McArthur, E.D., Stevens, R. 2004. Composite shrubs. In: Monsen, S.B., Stevens, R., Shaw, N.L., comps. *Restoring Western Ranges and Wildlands*. Gen. Tech. Rep. RMRS-GTR-136-vol. 2. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station: 493-538.
- McArthur, E.D., Taylor, J.R. 2004. *Artemisia filifolia* Torr: sandsage. In: Francis, J.K. ed. *Wildland Shrubs of the United States and Territories: Thamnisc Descriptions: Volume 1*. IITF-GTR-26. San Juan, PR: International Institute of Tropical Forestry, and Rocky Mountain Research Station: 60-62.
- McCune B., Grace, J.B. 2002. *Analysis of Ecological Communities*. MjM Software Design, Gleneden Beach, OR. 300 p.
- McLuckie, A.M., Bennion, M.R.M., Fridell, R.A., Radant, R. 2006. Status of the desert tortoise in the Red Cliffs Desert Reserve. Abstracts of the 2006 Desert Tortoise Council 31st Annual Meeting and Symposium, 2006 February 17-20, Tucson, AZ. [Online]. Available: <http://www.deserttortoise.org/abstract/abstracts2006/2006abstracts.pdf> [June 3, 2008].
- Mortensen, V.L., Carley, J.A., Crandall, G.C., Donaldson, K.M. Jr., Leishman, G.W. 1977. *Soil Survey of Washington County Area*, Utah. USDA, Soil Conservation Service, and USDI, Bureau of Land Management and National Park Service, in cooperation with Utah Agricultural Experiment Station. 140 p.
- NOAA (National Oceanic and Atmospheric Administration). 2008. St. George Utah Climatological Station, Summary Of The Month (SOM) Cooperative. Available: <http://www4.ncdc.noaa.gov/cgi-win/wcwg.dll?wDI-StnSrch-StnID~20026076> [May 14, 08].
- NRCS (USDA, Natural Resources Conservation Service). 2008. *Plants Database*. Available: <http://plants.usda.gov> [June 3, 2008].
- Oftedal, O.T. 2002. The nutritional ecology of the desert tortoise in the Mohave and Sonoran deserts. In: Van Devender, T.R., ed. *The Sonoran Desert Tortoise: Natural History, Biology and Conservation*. University of Arizona Press, Tucson, AZ: 194-241.
- Oftedal, O.T. 2005. Fast plants, slow tortoises: how nutrition could constrain the recovery of the desert tortoise. Abstracts of the 2005 Desert Tortoise Council 30th Annual Meeting and Symposium, 2005 February 18-20, Tucson, AZ. Available: <http://www.deserttortoise.org/abstract/abstracts2005/dtc2005abstracts.pdf> [June 3, 2008].
- Oftedal, O.T., Leuteritz, T.E.J. 2006. The distribution and abundance of high-PEP plants in the Mojave and Sonoran Deserts in a year of high rainfall. Abstracts of the 2006 Desert Tortoise Council 31st Annual Meeting and Symposium, 2006 February 17-20, Tucson, Available: <http://www.deserttortoise.org/abstract/abstracts2006/2006abstracts.pdf> [June 3, 2008].
- Pendleton, R.L., Pendleton, B.K., Howard, G.L., Warren, S.D. 2003. Growth and nutrient content of herbaceous seedlings associated with biological soil crusts. *Arid Land Research And Management* 17: 271-281.
- R Development Core Team. 2008. *R: A Language and Environment for Statistical Computing*, version 2.7.0. [Online]. Available: <http://www.R-project.org>
- Ramaley, F. 1939. Sand-hill vegetation of northeastern Colorado. *Ecological Monographs* 9: 1-51.
- Rasmussen, L.L., Brotherson, J.D. 1986. Habitat relationships of sandsage (*Artemisia filifolia*) in Southern Utah. In: McArthur, E.D., Welch, B.L., comps. *Proceedings—Symposium on the Biology of Artemisia and Chrysothamnus*, 1984 July 9-13, Provo, UT. Gen. Tech. Rep. INT-200. Ogden, UT: USDA, Forest Service, Intermountain Research Station: 58-66.
- Rose, L., Duck, T. 2001. Managing for Recovery in the Red Cliffs Desert Reserve. Abstracts of the 2001 Desert Tortoise Council 26th Annual Meeting and Symposium, 2001 March 16-18, Tucson, AZ. [Online]. Available: <http://www.deserttortoise.org/abstracts/abstracts2001/2001abs33.html> [June 3, 2008].
- Schiffman, P. M. 1994. Promotion of exotic weed establishment by endangered giant kangaroo rats (*Dipodomys ingens*) in a California grassland. *Biodiversity and Conservation* 3: 524-537.
- Sims, P.L., Dahl, B.E., Denham, A.H. 1976. Vegetation and livestock response at three grazing intensities on sandhill rangeland in eastern Colorado. Colorado Agricultural Experiment Station, Technical Bulletin 130. 48 p.
- Smith, M., Rose, S. 1996. U.S. Fish and Wildlife Service issues incidental take permit to Washington County, Utah, for desert tortoises. New Release, U.S. Fish and Wildlife Service. [Online]. Available: <http://www.fws.gov/mountain-prairie/pressrel/96-17.html> [June 3, 2008].
- Stevens, R. 2004. Basic considerations for range and wildland revegetation and restoration. In: Monsen, S.B., Stevens, R., Shaw, N.L., comps. *Restoring Western Ranges and Wildlands*. Gen. Tech. Rep. RMRS-GTR-136-vol 1. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station: 493-538.

- Thornburg, A.A. 1982. Plant materials for use on surface-mined lands in arid and semiarid regions. Tech. Rep. SCS-TP-157, EPA-600/7-79-134. Washington, DC: Soil Conservation Service, Superintendent of Documents, Government Printing Office. 88 p.
- USFWS (U.S. Fish and Wildlife Service). 1990. Endangered and threatened wildlife and plants, determination of threatened status for the Mojave population of the desert tortoise. Federal Register 55:12178-12191. Available: http://ecos.fws.gov/docs/federal_register/fr1673.pdf [July 22, 2008].
- USFWS (U.S. Fish and Wildlife Service). 1994. Endangered and threatened wildlife and plants, determination of critical habitat for the Mojave population of the desert tortoise. Federal Register 59:5820-5866. Available: http://ecos.fws.gov/docs/federal_register/fr2519.pdf [July 22, 2008].
- Webb, R.H., Steiger, J.W., Turner, R.M. 1987. Dynamics of Mojave Desert shrub assemblages in the Panamint Mountains, California. Ecology 68: 478-490.
- Welsh, S.L., Atwood, N.D., Goodrich, S., Higgins, L.C. 1987. A Utah Flora. Brigham Young University, Provo, Utah.
- Welsh, S.L., Atwood, N.D., Goodrich, S., Higgins, L.C. 2003. A Utah Flora, Third Ed., revised. Brigham Young University, Provo, Utah.
- WRCC (Western Regional Climate Center). Classification of El Niño and La Niña Winters. [Online]. Available: <http://www.wrcc.dri.edu/enso/ensodef.html> [June 3, 2008].

Converting Crested Wheatgrass Stands to Enhance Big Sagebrush: A Literature Review

Krystle A. Pehrson¹ and Bok F. Sowell²

ABSTRACT

Greater sage-grouse (*Centrocercus urophasianus*) is a high priority species for federal and state land management agencies. Sage-grouse are sagebrush (*Artemisia* spp.) obligates requiring sagebrush for their survival throughout the year. Sagebrush has been removed and replaced with crested wheatgrass (*Agropyron cristatum* & *A. desertorum*) throughout the West. The objectives of this paper were to review the literature (99 papers), as well as consult experts, to determine which methods are most likely to eliminate crested wheatgrass and establish sagebrush. No technique eliminates crested wheatgrass in a single application. Grazing and fire have no long-term impacts on crested wheatgrass. Mechanical treatments, such as plowing, disking, and cultivating reduce and eradicate crested wheatgrass, but a flush of invasive annual grasses following mechanical disturbance can make establishment of seeded species difficult. It appears that the best way to reduce crested wheatgrass cover and establish sagebrush is to spray crested wheatgrass with glyphosate in early spring for two consecutive years at a rate of 1.1 kg/ha of active ingredient. Then, sagebrush should be seeded in the late fall using a compact row seeder or Brillion cultipacker at a rate of 0.22 kg/ha pure live seed.

INTRODUCTION

Greater sage-grouse (*Centrocercus urophasianus*) is a high priority species for federal and state land management agencies. In 2005 there were three petitions to list greater sage-grouse as threatened or endangered under the Endangered Species Act of 1973 (U.S. Fish and Wildlife Service 2005). Breeding populations of sage-grouse have declined by 17 to 47 percent throughout much of their range during the last 70 years due to decreases in the quality and quantity of sagebrush habitats (Connelly and others 2000). Sage-grouse are sagebrush (*Artemisia* spp.) obligates requiring sagebrush for their survival throughout the year (Connelly and others 2000). Sagebrush has been removed and replaced with crested wheatgrass (*Agropyron cristatum* (L.) Gaertn. & *A. desertorum* (Fisch. ex Link) J.A. Schultes) throughout the West (Lorenz 1986). Crested wheatgrass stands form monocultures, with low sagebrush cover and understory plant species diversity (Heidinga and Wilson 2002, Henderson and Naeth 2005). The Bureau

of Land Management (BLM) in Montana has committed funds to restore crested wheatgrass to sagebrush to provide more suitable sage-grouse habitat (J. Parks personal communication, Sept 2007).

The objectives of this paper are to review the literature, as well as consult experts, to determine which methods are most likely to be successful to eliminate crested wheatgrass and establish sagebrush to improve sage-grouse habitat. Crested wheatgrass elimination techniques reviewed include grazing, burning, chemical and mechanical treatments. Sagebrush establishment techniques reviewed include drilling and broadcasting seed.

METHODS

This study was based on a review of 99 papers. Eighteen papers were government reports, seven were theses, seven were published in symposium proceedings, and the 67 remaining were peer-reviewed, scientific studies. Twenty-five studies dealt with sage-grouse habitat requirements, and the remaining 74 dealt specifically with modifying crested wheatgrass stands or establishing sagebrush. Four of the 74 crested wheatgrass studies were observational, but relied on long-term records to reach their conclusions. Results from the literature presented in this review were included if there was a significant *P*-value (0.05). Differences of opinion between papers were resolved by giving preference to long-term replicated studies with strong experimental designs. All 99 studies were reviewed to reach management recommendations.

In order to ensure that our conclusions and management recommendations were reasonable, we consulted seven experts experienced with crested wheatgrass and sagebrush establishment. Experts were chosen based on experience or volume of published reports. The experts included Mike Pellant, Idaho State BLM Great Basin Initiative Coordinator, Nancy Shaw, United States Forest Service (USFS) Research Botanist, Dr. Bruce Roundy, Brigham Young University Plant and Wildlife Sciences Professor, Dr. Jane Mangold, U.S. Department of Agriculture, Agriculture Research Station Ecologist, Steve Monsen, USFS Botanist (retired), Kyle Wendtland, Antelope Mine Powder River Basin, Wyoming, and Stuart Wengreen, Simplot Grower Solutions Certified Crop Advisor. There were no major discrepancies between the scientific literature and expert opinion. To be included in this review, corroboration by two or more experts was required. I also asked experts for information and advice not included in the scientific literature that would help land managers.

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Krystle A. Wengreen, Eastern Idaho Supervisory Area, Idaho Department of Lands, Idaho Falls, kwengreen@idl.idaho.gov

²Bok F. Sowell, Department of Animal and Range Sciences, Montana State University, Bozeman, MT

LITERATURE REVIEW

Sage-Grouse Habitat Requirements

Sage-grouse are a sagebrush obligate species and require large tracts of relatively continuous sagebrush cover (Connelly and others 2000). Sage-grouse require sagebrush throughout the entire year. Sagebrush comprised 62 percent of the year-long diet of adult sage grouse and essentially 100 percent of their winter diet in Montana (Wallestad and others 1975). Sage-grouse require a variety of shrubs in different quantities for wintering, breeding, nesting, and brood-rearing (Connelly and others 2000, Crawford and others 2004).

Sage-grouse are completely dependent on sagebrush communities throughout the winter (Wambolt and others 2002). During the winter, sage-grouse commonly use medium to tall sagebrush communities on south and west facing slopes (Connelly and others 2000), as well as sagebrush habitats with 12 (Woodward 2006) to 20 percent canopy cover in Montana (Wallestad and others 1975). Sage-grouse used areas with 1.2 shrubs/m² in central Montana (Woodward 2006). Sage-grouse feed almost exclusively on sagebrush throughout the winter, with big sagebrush (*A. tridentata* Nutt.) dominating their diet (Wallestad and others 1975). Other sagebrush species such as low sagebrush (*A. arbuscula* Nutt.), black sagebrush (*A. nova* A. Nels.), fringed sagebrush (*A. frigida* Willd.), and silver sagebrush (*A. cana* Pursh) are consumed depending on availability (Wambolt and others 2002).

Lekking or breeding habitat is typically located in open areas surrounded by sagebrush, which may include ridgetops, swales, dry lakebeds, burned areas, or plowed fields (Connelly and others 1981, Gates 1985). Sage-grouse males tend to form leks in areas that are within or adjacent to potential nesting habitat (Connelly and others 2000). Sagebrush, herbaceous plant cover, and insects are major components of the lek habitat (Connelly and others 2000).

During the pre-laying period, 50 to 80 percent of the hen's diet is sagebrush (Barnett and Crawford 1994, Crawford and others 2004). Pre-laying hens need a diversity of forbs high in calcium, phosphorus, and protein (Barnett and Crawford 1994). The quality of the pre-laying areas may impact nest initiation rate, clutch size, and future reproductive successes (Barnett and Crawford 1994). During the pre-laying period habitat use centers around low sagebrush and Wyoming big sagebrush (*A. tridentata* ssp. *wyomingensis* Rydb.; (Crawford and others 2004).

Sage-grouse require sagebrush stands with 15 to 38 percent sagebrush cover for nesting and almost always nest under sagebrush (Connelly and others 2000, Lane 2005, Sika 2006, Woodward 2006). In Montana 91 percent of sage-grouse nested under sagebrush, with less than two percent

nesting under crested wheatgrass (Lane 2005). In another study in Montana, 91 percent were located under sagebrush and six percent were located in either seeded grasslands (includes crested wheatgrass) or alfalfa fields (Sika 2006). Nest success is higher (53 percent) when sage-grouse are able to use sagebrush, compared with non-sagebrush nests (22 percent) (Connelly and others 1991). Sage-grouse nest selection is related to sagebrush height and cover (Crawford and others 2004, Klebenow 1969). Sage-grouse nest sites commonly occur in sagebrush stands with 15 to 38 percent sagebrush cover, 3 to 30 percent herbaceous plant cover and 14 to 34 cm grass heights (Connelly and others 2000). The average height of sagebrush associated with nests was between 29 to 80 cm, with nests commonly under the tallest sagebrush within a stand (Connelly and others 2000, Wallestad and Pyrah 1974). In central Montana, sage-grouse nest habitats had 15 percent shrub cover with an average height of 28 cm and a density of 1.4 shrubs/m² (Woodward 2006). Another study in Montana found that nesting habitat had shrub cover of 9 to 12 percent (Moynahan 2004). Sage-grouse depend on grasses and forbs in the understory, which may provide scent, visual, and physical barriers to potential predators (Delong and others 1995).

Brood-rearing areas are commonly in upland sagebrush habitat (Connelly and others 2000). Most are relatively open stands of sagebrush with 14 percent canopy cover and ≥15 percent herbaceous plant cover (Martin 1970, Sveum and others 1998, Wallestad 1971). Most (90 percent) brood sites are located in Wyoming big sagebrush while less than four percent are found in crested wheatgrass (Lane 2005). In central Montana, brood sites had 14 percent shrub cover with an average shrub height of 28 cm and a density of 1.4 shrubs/m² (Woodward 2006). Most brood areas are characterized by high plant species richness with abundant forbs and insects (Drut and others 1994a, Dunn and Braun 1986, Klott and Lindzey 1990). Diets of sage-grouse chicks included 34 genera of forbs and 41 families of invertebrates in Oregon (Drut and others 1994b). Forbs and insects comprise the majority of chick diets until they are 12 weeks old, when sagebrush becomes a major component (Klebenow and Gray 1968). As the growing season progresses, more mesic habitats, such as riparian areas, are used as forbs desiccate (Crawford and others 2004, Wallestad 1971).

Crested Wheatgrass History

Between 13 to 17 million ha of crested wheatgrass have been planted in North America (Bakker and Wilson 2004) since it was introduced from Russia in 1897 (Dillman 1946, Lorenz 1986). In the early 1900s, crested wheatgrass, a drought and cold resistant plant, quickly became the most commonly used introduced perennial grass species for revegetation in the northern Great Plains and much of the West (Laycock 1981, Rogler and Lorenz 1983, Sharp 1986). Crested wheatgrass was used to control erosion,

increase livestock forage, control weeds, and reduce wildfires (Lorenz 1986, Smoliak and Dormaar 1985, Young and Evans 1986). The first large scale plantings of crested wheatgrass was prompted by large sheep losses to halogeton (*Halogeton glomeratus* (Bieb.) C.A. Mey.) poisoning (Mathews 1986). Land managers needed a perennial plant that would suppress and crowd out halogeton (Mathews 1986).

Crested wheatgrass has been planted over a large portion of the historic and current range of sage-grouse (Connelly and others 2000, Rogler and Lorenz 1983, Schroeder and others 2004). Crested wheatgrass forms large homogenous stands, lacking the sagebrush and plant species diversity required by sage-grouse (Crawford and others 2004, Heidinga and Wilson 2002). Sagebrush cover is lower in crested wheatgrass stands when compared to native sagebrush grasslands (McAdoo and others 1989). Large crested wheatgrass stands become monocultures and displace native plant species, such as sagebrush, for 40 to 50 years (Hull and Klomp 1966, Looman and Heinrich 1973, Marlette and Anderson 1986). Where sagebrush is completely removed sage-grouse will cease to nest (Klebenow 1969). Dense crested wheatgrass stands reduce feed and structure for sage-grouse adults and chicks (Pellant and Lysne 2005) by excluding native grasses, shrubs, and forbs. Lower plant species diversity in crested wheatgrass result in reduced populations of invertebrates necessary for survival and growth of sage-grouse chicks (DeLuca and Lesica 1996). Increasing sagebrush within crested wheatgrass stands would improve habitat for sage-grouse throughout the year.

Crested Wheatgrass Conversion

Because crested wheatgrass competes for light, nutrients and water, it must be reduced or eliminated to facilitate the establishment of sagebrush (Pellant and Lysne 2005). The greatest obstacle to seedling establishment in any habitat is the vegetation currently established in that habitat (Grubb 1977). Control of herbaceous vegetation is necessary for successful sagebrush establishment, because grasses out compete young shrubs (Kiger and others 1987). Crested wheatgrass is a prolific seed producer and has a large, viable seed bank in the soil, which may affect the success of suppression or eradication strategies (Marlette and Anderson 1986, Pellant and Lysne 2005). We review the use of grazing, fire, mechanical and/or chemical treatments to determine the best method to reduce crested wheatgrass cover sufficient for sagebrush establishment.

Grazing

Crested wheatgrass can withstand heavy grazing for a number of years (Caldwell and others 1981, Cook and others 1958, Hull 1974, Hull and Klomp 1966, Laycock and Conrad 1981). Crested wheatgrass stands in Idaho were heavily grazed for over 20 years, but were still vigorous and showed no signs of deterioration (Hull and Klomp 1966).

Heavy use (up to 70 percent) alone has been shown to have little long-term impact on crested wheatgrass stands (Frischknecht and Harris 1968). Crested wheatgrass persisted after 20 years of annual heavy spring grazing (Hubbard 1949), even though grazing occurred when it should be most deleterious (Olson and Richards 1988). Mowing crested wheatgrass in the fall did not reduce the basal area compared to the control the following spring (Lodge 1960). When grazing pressure was removed, stands recovered quickly, doubling yields two seasons after grazing ceased (Hull and Klomp 1966).

Clipping may not have an impact on crested wheatgrass cover, but seed production can be reduced to zero (Cook and others 1958, Hansen and Wilson 2006). Growth may be reduced under cumulative, repetitive defoliation due to reduced root growth and thus reduce its ability to take up nutrients (Hansen and Wilson 2006). Even though seed production was reduced, it has been shown that overall crested wheatgrass cover is unaffected by long-term grazing (Caldwell and others 1981, Frischknecht and Harris 1968, Hull 1974, Hull and Klomp 1966, Laycock and Conrad 1981).

Burning

Burning crested wheatgrass has no long-term impacts on biomass production (Lodge 1960, Romo and others 1994). Fall burning to rejuvenate pastures reduced the yield of crested wheatgrass, but in the following years, production equaled or exceeded controls (Lodge 1960). Spring burning reduced crested wheatgrass production for two years, but returned to control levels after two years. The vegetative cover of crested wheatgrass was unchanged two seasons following spring or fall burns in two separate studies almost 35 years apart in Saskatchewan (Lodge 1960, Romo and others 1994). The cost of prescribed burning varies between approximately \$10.00 and \$80.00/ha depending on the size of the burn (Gee and Biermacher 2007). Because burning does not reduce crested wheatgrass cover, it is not a useful pretreatment for sagebrush establishment.

Mechanical Treatments

Mechanical treatments are effective in removing or reducing vigor in crested wheatgrass plants (Pellant and Lysne 2005). Common treatments include disking, rototilling, cultivating, and harrowing (Pellant and Lysne 2005).

In Canada, double disking reduced crested wheatgrass basal area 60 percent, but the treatment effect was not permanent (Lodge 1960). Rototilling or disking reduced crested wheatgrass cover by 72 to 87 percent, after which crested wheatgrass cover was 13 to 17 percent (Bakker and others 1997, Wilson and Gerry 1995). Spring plowing effectively eliminated crested wheatgrass for two years (Lorenz and Rogler 1962).

Two tillage treatments were applied for 2 consecutive years to reduce crested wheatgrass prior to planting native plant species in an arid (17.4 cm annual precipitation) portion of Utah (Cox and Anderson 2004). In one treatment, a cultivator completely removed all existing vegetation and mixed the soil to 18 cm. In the second, a harrow was used to uproot some, but not all crested wheatgrass plants. Cultivation reduced crested wheatgrass cover by 84 to 93 percent and harrowing reduced cover by 50 to 67 percent. Essentially no seedlings survived without a disturbance (tilling, harrowing, chemical application). Treatments created openings that persisted long enough to allow seeded native species to establish.

The effectiveness of mechanical treatments varies greatly and problems with increased soil erosion and weed entry may impact their usefulness for establishing native plant species (Pellant and Lysne 2005). The drawbacks associated with mechanical treatments include the increased susceptibility to wind and water erosion, loss of the soil structure, disruption of soil moisture and nutrient cycling, and the increased risk for establishment of undesirable species (Bakker 1996). Facilitation of topsoil erosion is of particular concern when treatments result in complete removal of crested wheatgrass (Lorenz and Rogler 1962). Tillage also damages desirable, native plants that may be present within crested wheatgrass stands.

A flush of invasive annual grasses following mechanical disturbance can make establishment of seeded species difficult (King and others 1989). In the Great Basin, cheatgrass (*Bromus tectorum* L.) invades sites that have been mechanically disturbed making establishment of sagebrush difficult (M. Pellant, personal communication, October 2007). Cheatgrass is an introduced annual grass that dominates 1.3 million ha of public lands in the Great Basin and another 30.8 million ha are classified as either infested or susceptible to cheatgrass invasion (Pellant and Hall 1994). Annual weedy species such as cheatgrass reduce the success of sagebrush establishment (Cook 1974). In Utah, cheatgrass cover was not affected by tilling, harrowing, or herbicide applications (Cox and Anderson 2004). It is crucial to know if invasion by cheatgrass or other invasive weeds is a risk before mechanical treatments are performed.

Prices of various mechanical treatments range from \$59/ha to plow, \$39/ha to disk and \$35/ha to harrow, which are based on custom farming rates (S. Wengreen, personal communication, October 2007). Mechanical treatments are effective in reducing crested wheatgrass cover enough to establish sagebrush, but may not be worth the risk of invasion by weedy annual grasses like cheatgrass.

Chemical Treatments

The use of appropriate herbicides can help control crested wheatgrass and thus improve the opportunity for establishment of seeded species (Whisenant 1999). Application of glyphosate (*N*-[phosphonomethyl]glycine), a generalist herbicide, has been shown to be effective in reducing vigor or causing mortality of crested wheatgrass (Bakker and others 2003, Romo and others 1994, Wilson and Gerry 1995, Wilson and Partel 2003). Glyphosate applications reduce growth of treated crested wheatgrass plants for about eight weeks, providing a period of reduced competition (Gobin 1994). Herbicide application at the correct phenological stage (8 to 15 cm height) is crucial for determining treatment effectiveness. Applications in the early spring when crested wheatgrass is beginning growth and before desired native species initiate growth, appears to be the most effective (Ambrose and Wilson 2003, Bakker and others 1997, Bakker and others 2003, Peat and Bowes 1995, Romo and others 1994). Glyphosate applications should not be relied upon to completely eliminate crested wheatgrass (Cox and Anderson 2004, Romo and others 1994, Wilson and Partel 2003).

Cox and Anderson (2004) found that two herbicide applications reduced crested wheatgrass cover by 2 to 77 percent in Utah. Sagebrush and rabbitbrush (*Ericameria nauseosa* (Pallas ex Pursh) Nesom & Baird) were then seeded into treated plots using three methods (Cox and Anderson 2004). Crested wheatgrass cover was reduced to 4 percent the first year and 14 percent the second year. There was little to no establishment of sagebrush or rabbitbrush when crested wheatgrass cover was above 14 percent. Thus 14 percent provides an approximate threshold for the maximum crested wheatgrass cover allowed for establishment of shrubs in crested wheatgrass stands (Cox and Anderson 2004). Sagebrush and rabbitbrush emergence was 1.5 plants/m² in drilled plots and 0.6 plants/m² in broadcast and covered plots when glyphosate was applied in early spring for two consecutive years prior to planting (Cox and Anderson 2004). Tilling the seedbed prior to drilling resulted in 2.1 plants/m², while harrowing prior to drilling resulted in 1.1 plant/m². Although the tillage/drill treatment combination resulted in the highest shrub establishment, even the lower density of 0.6 plants/m² would be an acceptable density if these shrubs fully established. Plots were reexamined in the second growing season to assess establishment. Tilled and drilled plots resulted in 3.9 shrubs/m² and harrow and drill plots resulted in 1.8 shrubs/m². Herbicide and drilled plots resulted in 2.1 shrubs/m² and broadcast and cover plots yielded 0.9 shrubs/m². Establishment data is consistent with emergence data, where tilled plots had the highest rates of shrub establishment. The second year of the study had little emergence or establishment of any seeded plant species due to a lack of precipitation. During the first year there was above average precipitation in February and March which facilitated the establishment of seeded species (Cox and Anderson 2004).

Glyphosate applications are useful in reducing the cover of crested wheatgrass, especially after two early spring applications (Bakker and others 1997, Bakker and others 2003, Cox and Anderson 2004, Johnson 2004). A twice repeated spring application of glyphosate in Saskatchewan reduced crested wheatgrass yields by 77 percent (Gobin 1994). Another study in Saskatchewan applied glyphosate in early spring for two consecutive years and mean crested wheatgrass cover was reduced from 5 to 36 percent (Wilson and Gerry 1995). Cover of crested wheatgrass was reduced by 5 to 93 percent canopy cover in chemical treated plots compared to control plots after seven years of herbicide applications (Wilson and Partel 2003). In all of these studies crested wheatgrass cover was reduced below the required threshold of 14 percent for successful sagebrush establishment.

Although glyphosate does not completely eradicate crested wheatgrass, it works well for suppressing crested wheatgrass enough to allow for seeded plant species to have a competitive advantage for establishment. Treatments should be repeated, since crested wheatgrass may recover following treatment (Hansen and Wilson 2006). Applying glyphosate for two consecutive springs appears to reduce crested wheatgrass cover enough to establish sagebrush (Cox and Anderson 2004). Because drought years have a large impact on the establishment, seed may have to be applied a few times if there are years of below average precipitation (Cox and Anderson 2004, Johnson 2004). The price of spraying herbicide at a rate of 1.1 kg/ha of active ingredient (Bakker and others 1997, Bakker and others 2003, Hansen and Wilson 2006, Wilson and Partel 2003) is approximately \$34/ha (S. Wengreen, personal communication, October 2007). Herbicide applications do not cause the large scale soil disturbances that mechanical treatments do, making them a better option for avoiding weed invasion (M. Pellant, personal communication, October 2007).

Sagebrush Establishment

Sagebrush can be seeded once crested wheatgrass cover has been reduced to below 14 percent (Cox and Anderson 2004). Desired plant species, such as sagebrush or other native plants, must be planted as seeds or as bare root stock. Successful sagebrush seedlings should result in a shrub density of at least 1.2 to 1.4 shrubs/m². This is an average based on the areas occupied by sage-grouse during nesting, brood-rearing, and wintering in central Montana (Woodward 2006). Typically, long term survival rates of seeded sagebrush are less than 60 percent (Schuman and Belden 2002, Kiger and others 1987), so it may not be possible to fully evaluate seeding success until 8 or more years after initial seeding (Schuman and others 2005).

Seedbed Ecology and Preparation

Precipitation can negatively or positively influence species composition, productivity, survival, and establishment (Bleak and others 1965, Lauenroth and others 1994, Sala and others 1988, Weaver 1950). Climatic conditions strongly influence establishment even with proper site preparation and seeding methods (Call and Roundy 1991, Cox and Anderson 2004, Johnson 2004). Establishment of shrubs such as sagebrush can be difficult. Many failures are due to inadequate soil moisture at the time of germination and during the initial growth period (Holmgren 1956). Drought is the main cause of sagebrush seedling mortality (Meyer 1992). Late fall or early winter are the best times to plant sagebrush, as this is when sagebrush seed disperses naturally and soil surfaces are likely to be moist (Shaw and others 2005). Seedlings have been most successful when done in the late fall or early winter just prior to snowfall (Lambert 2005, Monsen 2000, Stevens and Monsen 2004). Sagebrush seeding in the spring is not recommended (Meyer 1992). Sagebrush seed requires 20 to 30 days of moisture to germinate; surface moisture within the top 0.2 to 0.5 cm of soil is critical (S.B. Monsen, personal communication, October 2007).

Exploiting weather conditions can help improve sagebrush seeding success (S.B. Monsen, personal communication, October 2007; M. Pellant, personal communication, October 2007; K. Wendtland, personal communication, October 2007). In the Powder River Basin of Wyoming, seeding sagebrush prior to storms in the late fall provides the best opportunity for establishment (K. Wendtland, personal communication, October 2007). The amount and timing of winter snowfall has a large impact on sagebrush seedling emergence. A single late-winter snowfall made the difference between zero percent emergence at one site and 80 percent emergence at a similar site in Wyoming (Monsen and Meyer 1990). Emergence is rarely achieved unless winter snow is present (Meyer 1992).

Big sagebrush should be planted on a firm seedbed with only a light covering of soil. Rough seedbeds may bury the seed too deeply (Shaw and others 2005). Providing a very firm seedbed is crucial for sagebrush establishment, so it may be necessary to use a cultipacker or some machinery to pack down the soil (K. Wendtland, personal communication, October 2007). It is crucial that seed not be buried too deeply (Monsen 2000, Lambert 2005). Site preparation or planting methods that create a loose, sloughing seedbed should be avoided (Meyer 1992).

Sagebrush Seeding Rates

Suggested seeding rates for basin big sagebrush (*A. tridentata* Nutt. ssp. *tridentata*), Wyoming big sagebrush and mountain big sagebrush (*A. tridentata* Nutt. ssp. *vaseyana* (Rydb.) Beetle) vary depending on the type of project. Mine reclamations use a larger amount of seed when compared to large-scale rangeland projects.

Sagebrush seeding rates for mine reclamation projects range from 1 to 4 kg/ha pure live seed (PLS) (Hild and others 2006, Schuman and Belden 2002, Schuman and others 1998, Schuman and others 2005, Vickland and others 2004). Large-scale rangeland projects typically use a much lower seeding rates, ranging from 0.05 to 0.56 kg/ha PLS (Dalzell 2004, Lambert 2005, Meyer 1992, Monsen 2000, Shaw and others 2005). A typical seeding rate when drilling sagebrush ranges from 0.05 to 0.22 kg/ha PLS (Lambert 2005, Meyer 1992, Monsen 2000, Shaw and others 2005). Broadcast seeding normally requires 33 to 50 percent more seed to be used compared to drilling, with seeding rates ranging from 0.22 to 0.56 kg/ha PLS (Dalzell 2004, Meyer 1992, Monsen 2000, Stevens and Monsen 2004). We recommend a seeding rate of 0.22 kg/ha PLS when drilling, and a rate of 0.33 kg/ha PLS when broadcast seeding.

The current market price of sagebrush seed may dictate the amount of seed used. Sagebrush seed ranges from \$70 to \$300/kg PLS depending on the current market price and availability (Schuman and others 2005). Current market prices (2007) for big sagebrush subspecies seed range from \$101 to 168/kg PLS (K. Wengreen, personal communication, October 2007). A midrange price of \$135/kg PLS will be used for economic analysis in this paper.

Drilling

The rangeland drill or other single disk drills can be used to plant sagebrush seeds. Drilling commonly requires 33 to 50 percent less seed than broadcast seeding (Stevens and Monsen 2004). Big sagebrush may be seeded through drills if seed is dropped on or near the soil surface then covered lightly by press wheels or by pulling a cultivator behind the drill (Lambert 2005, Shaw and others 2005). The seed drill should be set to place the sagebrush seed 0.2 cm into the soil (Lambert 2005), placing seeds deeper than 0.6 cm reduces emergence (Monsen 2000).

In Nevada, sagebrush was seeded using a traditional rangeland drill to a depth of 0.6 to 1.3 cm and only 1.2 seedlings/m² established, while leaving the seed on the soil surface resulted in 99 to 101 seedling/m² (Monsen and Meyer 1990). In Utah, sagebrush and rabbitbrush were seeded using a Truax rangeland drill and John Deere Flex Drill set to a depth of 1 cm and 1.4 to 3.9 seedling/m² established and were present the following year (Cox and Anderson 2004). Drill seeding sagebrush and rabbitbrush in crested wheatgrass following cultivation resulted in 50 percent more emergence (2.1 seedlings/m²) than broadcast seeding only and broadcast seeding followed by covering (Cox and Anderson 2004).

Many devices have been used to compact the soil surface, resulting in increased sagebrush seedling establishment. These devices are not necessarily drills, but utilize a tractor to go over the soil surface, which is why they have been included in this section. The Oyer compact row seeder

compacts the soil then presses the seed into the soil surface (Lysne 2005). In Nevada, use of the Oyer compact row seeder resulted in very high initial sagebrush seedling density (646 seedlings/m²) and 1-year survival (101 seedlings/m²; Monsen and Meyer 1990). A second implement, the Brillion cultipacker seeder broadcasts the seed over the soil surface then presses it into the soil (Lysne 2005). In Nevada, use of the Brillion cultipacker resulted in emergence of 352 sagebrush seedlings/m² and 1-year survival of 99 seedling/m² (Monsen and Meyer 1990). A Truax Rough Rider drill with brillion wheels was recommended by 3 experts that have experience with establishing sagebrush within crested wheatgrass stands (J. Mangold, personal communication, October 2007; M. Pellant, personal communication, October 2007; N. Shaw, personal communication, October 2007). According to preliminary results, the Truax Rough Rider drill with brillion wheels resulted in sagebrush establishment of 1.3 seedlings/m² (N. Shaw, personal communication, October 2007).

The cost of drilling seed is highly dependent on the amount of sagebrush seed used and can range between \$25 to 40/ha (M. Pellant, personal communication, October 2007). Using a drill at a rate of \$33/ha, and a heavy seeding rate of 0.22 kg/ha PLS would be approximately \$63/ha. Drilling sagebrush seed has resulted in sagebrush establishment densities to meet the average found in areas occupied by sage-grouse in central Montana (Woodward 2006).

Broadcast Seeding

Conventional tilling and seeding methods may be impractical on some rangelands due to rocky or steep terrain. These sites require land managers to utilize broadcast seeding. Large areas can be seeded with sagebrush quickly using aircraft (Monsen 2000). Aerial broadcasting is often easier than drilling seed, since topography is not a limiting factor (Monsen 2000). Broadcast seeding helps to keep seeds near the soil surface, which is important for planting sagebrush (Shaw and others 2005).

In Idaho, big sagebrush failed to establish on 23 of 25 fire rehabilitation projects when aerially seeding was used without seedbed preparation (Dalzell 2004). In Utah, broadcast seeding followed by covering the seed resulted in 46 percent more sagebrush and rabbitbrush seedlings than broadcast seeding alone (Cox and Anderson 2004). Coverage of broadcast seed using chains, harrows, rails, or other implements is recommended to ensure that seed comes into contact with soil surface (Stevens and Monsen 2004). Light anchor chaining increased seedling density at Dry Creek Drainage, Idaho from approximately 0.6 shrubs/m² on unchained sites to 6.4 shrubs/m² on chained sites with similar south and west aspects (Monsen 2000). Chaining increased seedling establishment and produced more uniform stands. On a mine reclamation project in

Wyoming, sagebrush seed was broadcast seeded using a precision broadcast seeder with cracked corn to accomplish uniform seed distribution and flow through the seeder. Seeding rates between 1 to 4 kg/ha PLS were used and had sagebrush establishment densities of 1.0 to 4.5 shrubs/m² after 6 years (Williams and others 2002).

The price of broadcasting sagebrush seed is highly dependent on the amount of sagebrush seed used. The cost of broadcasting depends on the method used. Aerial broadcasting costs \$12 to 19/ha, while broadcasting on the ground costs \$25 to 40/ha to apply (M. Pellant, personal communication, October 2007). Using an aerial application at a rate of \$16/ha and a heavy seeding rate of 0.33 kg/ha PLS would cost approximately \$61/ha.

Interseeding and Transplanting

Interseeders and transplanters are another category of mechanical equipment that may be used to remove competition in narrow bands while planting shrubs in a one-pass operation (Giunta and others 1975, Stevens 1992, Stevens and others 1981). Two methods for removing vegetation prior to interseeding or transplanting include scalping or herbicide (Stevens 1992). Scalping is done using modified disks or a plow pulled behind a tractor; it must be wide and deep enough to remove or kill all seeds or rhizomes of existing vegetation (Stevens 1992). Existing vegetation can also be killed in strips or spots with herbicide. Sprayed areas can then be seeded by drill or broadcast seeding (Stevens 1992). Scalping or herbicide gives seeded plants a chance to become established before competing vegetation reinvades (Stevens 1992). The width of the scalp or herbicide spray area depends on the density, vigor, and growth form of the major species in the existing vegetation, the species to be interseeded and site conditions (Stevens 1992). Results from interseeding and transplanting are variable depending on local conditions, use of proper techniques, and plant materials used (Stevens 2004).

Interseeding techniques can be used to plant seedlings mechanically in areas that have been scalped or sprayed with herbicide. Scalped areas should be at least 61 cm wide when planting shrubs (Giunta and others 1975). One study suggested that scalp width should be at least 100 cm wide, an optimum width of 150 cm (Van Epps and McKell 1978).

Sagebrush seedlings (12 to 20 cm tall) have been planted in early spring using field-grown bare root stock or greenhouse-grown containerized plants (Lambert 2005). Springtime is optimal for transplanting because soil moisture is high, temperatures are low, and chance of rainfall is high (Stevens 2004). Transplanting bare root stock may be superior to container-grown stock (Stevens and others 1981). First year survival rates are commonly 80 percent or higher (Lysne 2005), although losses following the first year can range from 46 to 69 percent (Stevens and others 1981). It is recommended that sagebrush seedlings

be randomly placed in clumps or blocks on the best sites within the restoration area. Typically, only small critical areas are planted because of the high cost of using planting stock (Lysne 2005). These sites will become fertile islands of big sagebrush as they mature, and provide a seed source for natural regeneration (Lambert 2005). However, natural regeneration would take many years since sagebrush seeds are very small (0.018 g/100 seeds to 0.025 g/100 seeds depending on variety) and maximum dispersal distances are in the range of 30 m, but 85 to 90 percent fall within 1 m of the canopy (Young and Evans 1989).

The cost of bare root stock or greenhouse-grown containerized plants of Wyoming big sagebrush can be greater than \$2/seedling, therefore to achieve the 1 shrub/m² density would require a minimum of 2,000 seedlings at a cost of \$4,000/ha (Schuman and others 2005). Large-scale plantings would not be cost-effective across all rangelands, making this a viable option only for small restoration projects.

CONCLUSIONS

Crested wheatgrass forms large homogenous stands lacking the sagebrush and plant species diversity required by sage-grouse (Crawford and others 2004, Heidinga and Wilson 2002). The purpose of this paper was to determine the best methods to suppress crested wheatgrass in order to establish sagebrush to enhance these areas for sage-grouse based on the current scientific literature. Crested wheatgrass cover must be reduced for sagebrush to be seeded and successfully established, thus improving areas for sage-grouse habitat. The presence of sagebrush is the best predictor for sage-grouse use. No technique has proven to eliminate crested wheatgrass in a single application. Repeated treatments or combinations of treatments may be necessary to reduce crested wheatgrass biomass and increase the establishment of seeded species.

Sagebrush establishment is most successful when using a drill with a seeding rate of at least 0.22 kg/ha PLS (Lambert 2005, Monsen and Meyer 1990, Meyer 1992, Monsen 2000, Shaw and others 2005). According to S. Monsen (personal communication 2007), the method of seeding sagebrush is not a crucial factor, however, it is critical that sagebrush seed is not buried too deeply. This means that you can probably drill or broadcast seed sagebrush, although higher establishment rates have been achieved using drills with light soil covering.

Achieving a shrub density of 1 shrub/m² would be acceptable, however 1.2 to 1.4 shrubs/m² would provide better habitat for sage-grouse (Woodward 2006). It may not be possible to evaluate seeding success until at least eight years after initial seeding (Schuman and others 2005).

REFERENCES

- Ambrose, L.G., Wilson, S.D. 2003. Emergence of the introduced grass *Agropyron cristatum* and the native grass *Bouteloua gracilis* in a mixed-grass prairie restoration. *Restoration Ecology* 11: 110-115.
- Bakker, J.D. 1996. Competition and the establishment of native grasses in crested wheatgrass fields [Thesis]. Regina, Saskatchewan: University of Regina. 70 p.
- Bakker, J.D., Christian, J., Wilson, S.D., Waddington, J. 1997. Seeding blue grama in old crested wheatgrass fields in southwestern Saskatchewan. *Journal of Range Management* 50:156-159.
- Bakker, J.D., Wilson, S.D. 2004. Using ecological restoration to constrain biological invasion. *Journal of Applied Ecology* 41:1058-1064.
- Bakker, J.D., Wilson, S.D., Christian, J.M., Li X.D., Ambrose, L.G., Waddington, J. 2003. Contingency of grassland restoration on year, site, and competition from introduced grasses. *Ecological Applications* 13:137-153.
- Barnett, J.K., Crawford, J.A. 1994. Prelaying nutrition of sage grouse hens in Oregon. *Journal of Range Management* 47: 114-118.
- Bleak, A.T., Frishknecht, N.C., Plummer, A.P., Eckert, R.A. 1965. Problems in artificial and natural revegetation of the arid shadscale vegetation zone of Utah and Nevada. *Journal of Range Management* 18:59-68.
- Caldwell, M.M., Richards, J.H., Johnson, D.A., Nowak, R.S., Dzurec, R.S. 1981. Coping with herbivory: photosynthetic capacity and resource allocation in two semiarid *Agropyron* bunchgrasses. *Oecologia* 50:14-24.
- Call, C.A., Roundy, B.A. 1991. Perspectives and processes in revegetation of arid and semiarid rangelands. *Journal of Range Management* 44:543-549.
- Connelly, J.W., Arthur, W.J., Markham, O.D. 1981. Sage grouse leks on recently disturbed sites. *Journal of Range Management* 34:153-154.
- Connelly, J.W., Schroeder, M.A., Sands, A.R., Braun, C.E. 2000. Guidelines to manage sage grouse populations and their habitats. *Wildlife Society Bulletin* 28:967-985.
- Connelly, J.W., Wakkinen, W.L., Apa, A.D., Reese, K.P. 1991. Sage grouse use of nest sites in southeastern Idaho. *Journal of Wildlife Management* 55:521-524.
- Cook, C.W. 1974. Surface rehabilitation of land disturbances resulting from oil shale development: final report. Fort Collins, CO: Environmental Resources Center. p. 123.
- Cook, C.W., Stoddart, L.A., Kinsinger, F.E. 1958. Responses of crested wheatgrass to various clipping treatments. *Ecological Monographs* 28:237-272.
- Cox, R.D., Anderson, V.J. 2004. Increasing native diversity of cheatgrass-dominated rangeland through assisted succession. *Journal of Range Management* 57:203-210.
- Crawford, J.A., Olson, R.A., West, N.E., Mosley, J.C., Schroeder, M.A., Whitson, T.D., Miller, R.F., Gregg, M.A., Boyd, C.S. 2004. Ecology and management of sage-grouse and sage-grouse habitat. *Journal of Range Management* 57:2-19.
- Dalzell, C.R. 2004. Post-fire establishment of vegetative communities following reseeding on south Idaho's Snake River Plain. [Thesis]. Boise, ID: Boise State University. 112 p.
- Delong, A.K., Crawford, J.A., Delong, D.C. 1995. Relationships between vegetational structure and predation of artificial sage grouse nests. *Journal of Wildlife Management* 59:88-92.
- Deluca, T.H., Lesica, P. 1996. Long term harmful effects of crested wheatgrass on Great Plains grassland ecosystems. *Journal of Soil and Water Conservation* 51:408-409.
- Dillman, A.C. 1946. The beginnings of crested wheatgrass in North America. *Journal of American Society of Agronomy* 38:237-250.
- Drut, M.S., Crawford, J.A., Gregg, M.A. 1994a. Brood habitat use by sage grouse in Oregon. *Great Basin Naturalist* 54:170-176.
- Drut, M.S., Pyle, W.H., Crawford, J.A. 1994b. Diets and food selection of sage grouse chicks in Oregon. *Journal of Range Management* 47:90-93.
- Dunn, P.O., Braun, C.E. 1986. Summer habitat use by adult female and juvenile sage grouse. *Journal of Wildlife Management* 50: 228-235.
- Frischknecht, N.C., Harris, L.E. 1968. Grazing intensities and systems on crested wheatgrass in central Utah: response of vegetation and cattle. Tech Bulletin 1388., Washington, DC: USDA, Forest Service. 47 p.
- Gates, R.J. 1985. Observations on the formation of a sage grouse lek. *Wilson Bulletin* 97:219-221.
- Gee, K., Biermacher, J. 2007. Prescribed burning: what is the cost? Ag News., Ardmore, OK: The Samuel Roberts Nobel Foundation. p. 2.
- Giunta, B.C., Christensen, D.R., Monsen, S.B. 1975. Interseeding shrubs in cheatgrass with a browse seeder-scalper. *Journal of Range Management* 28:398-402.
- Gobin, S.M. 1994. Evaluation of grass establishment, development, and survival under sod conditions in the dry subhumid prairies. [Thesis]. Winnipeg, Manitoba: University of Manitoba. 171 p.
- Grubb, P.J. 1977. Maintenance of species-richness in plant communities - importance of regeneration niche. *Biological Reviews of the Cambridge Philosophical Society* 52:107-145.
- Hansen, M.J., Wilson, S.D. 2006. Is management of an invasive grass *Agropyron cristatum* contingent on environmental variation? *Journal of Applied Ecology* 43:269-280.
- Heidinga, L., Wilson, S.D. 2002. The impact of an invading alien grass (*Agropyron cristatum*) on species turnover in native prairie. *Diversity and Distributions* 8:249-258.
- Henderson, D.C., Naeth, M.A. 2005. Multi-scale impacts of crested wheatgrass invasion in mixed-grass prairie. *Biological Invasions* 7:639-650.
- Hild, A.L., Schuman, G.E., Vicklund, L.E., Williams, M.I. 2006. Canopy growth and density of Wyoming big sagebrush sown with cool-season perennial grasses. *Arid Land Research and Management* 20:183-194.
- Holmgren, R.C. 1956. Competition between annuals and young bitterbrush (*Purshia tridentata*) in Idaho. *Ecology* 37:370-377.
- Hubbard, W.A. 1949. Results of studies of crested wheatgrass. *Scientific Agriculture* 29:385-395.
- Hull, A.C. 1974. Species for seeding arid rangeland in southern Idaho. *Journal of Range Management* 27:216-218.
- Hull, A.C., Klomp, G.J. 1966. Longevity of crested wheatgrass in the sagebrush-grass type in southern Idaho. *Journal of Range Management* 19:5-11.
- Kiger, R.O., Berg, W.A., Herron, J.T., Phillips, C.M., Atkinson, R.G. 1987. Shrub establishment in the mountain shrub zone. In: Munshower, F.F. [ED.], Proc. 4th Biennial Symposium on Surface Mining and Reclamation of the Great Plains. 17-19 March 1987, Billings, MT: Montana State University, Reclamation Research Unit 8704. p. 132-140.
- King, M.A., Waller, S.S., Moser, L.E., Stubbendieck, J.L. 1989. Seedbed effects on grass establishment on abandoned Nebraska Sandhills cropland. *Journal of Range Management* 42:183-187.
- Klebenow, D.A. 1969. Sage grouse nesting and brood habitat in Idaho. *Journal of Wildlife Management* 33:649-662.
- Klebenow, D.A., Gray, G.M. 1968. Food habits of juvenile sage grouse. *Journal of Range Management* 21:80-83.
- Klott, J.H., Lindzey, F.G. 1990. Brood habitats of sympatric sage grouse and Columbian sharp-tailed grouse in Wyoming. *Journal of Wildlife Management* 54:84-88.

- Lambert, S.M. 2005. Seeding considerations in restoring big sagebrush habitat. *In*: Shaw, N.L., Pellant, M. and Monsen, S.B. [EDS.]. Sage-grouse habitat restoration. RMRS-P-38., Boise, ID: USDA, Forest Service, Rocky Mtn. Research Station. p. 75-80.
- Lane, V.R. 2005. Sage-grouse (*Centrocercus urophasianus*) nest and brood-rearing sagebrush habitat characteristics in Montana and Wyoming [Thesis]. Bozeman, Montana State University. 63 p.
- Lauenroth, W.K., Sala, O.E., Coffin, D.P., Kirchner, T.B. 1994. The importance of soil water in the recruitment of *Bouteloua gracilis* in the shortgrass steppe. *Ecological Applications* 4:741-749.
- Laycock, W.A. 1981. Crested wheatgrass. *Rangelands* 3:151-153.
- Laycock, W.A., Conrad, P.W. 1981. Responses of vegetation and cattle to various systems of grazing on seeded and native mountain rangelands in eastern Utah. *Journal of Range Management* 34: 52-58.
- Lodge, R.W. 1960. Effects of burning, cultivating, and mowing on the yield and consumption of crested wheatgrass. *Journal of Range Management* 13:318-321.
- Looman, J., Heinrich, D. 1973. Stability of crested wheatgrass pastures under long-term pasture use. *Canadian Journal of Plant Science* 53:501-506.
- Lorenz, R.J. 1986. Introduction and early use of crested wheatgrass in the Northern Great Plains. *In*: Johnson, K.L. (ED.), *Crested wheatgrass: its values, problems, and myths*, symposium proceeding, Logan, UT: Utah State University. p. 1-348.
- Lorenz, R.J., Rogler, G.A. 1962. A comparison of methods of renovating old stands of crested wheatgrass. *Journal of Range Management* 15:215-219.
- Lysne, C.R. 2005. Restoring Wyoming big sagebrush. *In*: Shaw, N.L., Pellant, M. and Monsen, S.B. [EDS.]. Sage-grouse habitat restoration. RMRS-P-38., Boise, ID: USDA, Forest Service, Rocky Mountain Research Station. p. 90-98.
- Marlette, G.M., Anderson, J.E. 1986. Seed banks and propagule dispersal in crested wheatgrass stands. *Journal of Applied Ecology* 23:161-175.
- Martin, N.S. 1970. Sagebrush control related to habitat and sage grouse occurrence. *Journal of Wildlife Management* 34:313-320.
- Mathews, W.L. 1986. Early use of crested wheatgrass seedlings in halogeton control. *In*: Johnson, K.L. [ED.], *Proc. Crested Wheatgrass: Its Values, Problems, and Myths*. 3-7 October 1986, Logan, UT: Utah State University. p. 27-28.
- Mcadoo, J., Longland, W.S., Evans, R.A. 1989. Nongame bird community responses to sagebrush invasion of crested wheatgrass seedlings. *Journal of Wildlife Management* 53:494-502.
- Meyer, S.E. 1992. Germination and establishment ecology of big sagebrush: implications for community restoration. *In*: Monsen, S.B., Kitchen, S.G. [EDS.]. *Ecology and management of annual rangelands*. General Technical Report INT-GTR-313., Ogden, UT: Forest Service, Intermountain Research Station. p. 244-251.
- Monsen, S.B. 2000. Establishment of big sagebrush (*Artemisia tridentata*) in semiarid environments. *In*: Entwistle, P.G., DeBolt, A.M., Kaltenecker, J.H., Steenhof, K. [EDS.]. *Sagebrush steppe ecosystems*. 21-23 June 1999, Boise, ID: USDI, Bureau of Land Management. p. 81-88.
- Monsen, S.B., Meyer, S.E. 1990. Seeding equipment effects on establishment of big sagebrush on mine disturbances. *In*: Munshower, F.F. [ED.], *Proc. 5th Billings Symposium on Disturbed Land Rehabilitation*. 25-30 March 1990, Billings, MT: Montana State University Reclamation Research Unit Publication No. 9003. p. 192-199.
- Moynahan, B.J. 2004. Landscape-scale factors affecting population dynamics of greater sage-grouse (*Centrocercus urophasianus*) in north-central Montana, 2001-2004 [Thesis]. Missoula, MT: University of Montana. 133 p.
- Olson, B.E., Richards, J.H. 1988. Tussock regrowth after grazing: intercalary meristem and axillary bud activity of tillers of *Agropyron desertorum*. *Oikos* 51:374-382.
- Peat, H.C., G.G. Bowes. 1995. Direct seeding into crested wheatgrass: final report. Project 90000088. Saskatchewan Agriculture and Food.
- Pellant, M., Hall, C. 1994. Distribution of two exotic grasses on public lands in the Great Basin: status in 1992. *In*: Monsen, S.B., Kitchen, S.G. [EDS.]. *Ecology and management of annual rangelands*. General Technical Report INT-GTR-313., Ogden, UT: Forest Service, Intermountain Research Station. p. 300-306.
- Pellant, M., Lysne, C.R. 2005. Strategies to enhance plant structure and diversity in crested wheatgrass seedlings. *In*: Shaw, N.L., Pellant, M. and Monsen, S.B. [EDS.]. Sage-grouse habitat restoration. RMRS-P-38., Boise, ID: USDA, Forest Service, Rocky Mountain Research Station. p. 81-92.
- Rogler, G.A., Lorenz, R.J. 1983. Crested wheatgrass: early history in the United States. *Journal of Range Management* 36:91-93.
- Romo, J.T., Grilz, P.L., Delanoy, L. 1994. Selective control of crested wheatgrass (*Agropyron cristatum* L Gaertn and *Agropyron desertorum* Fisch) in the Northern Great-Plains. *Natural Areas Journal* 14:308-309.
- Sala, O.E., Parton, W.J., Joyce, L.A., Lauenroth, W.K. 1988. Primary production of the central grassland region of the United States. *Ecology* 69:40-45.
- Schroeder, M.A., Aldridge, C.L., Apa, A.D., Bohn, J.R., Braun, C.E., Bunnell, S.D., Connelly, J.W., Deibert, P.A., Gardner, S.C., Hilliard, M.A., Kobriger, G.D., Mcadam, S.M., McCarthy, C.W., McCarthy, J.J., Mitchell, D.L., Rickerson, E.V., Stiver, S.J. 2004. Distribution of sage-grouse in North America. *The Condor* 106:363-376.
- Schuman, G.E., Belden, S.E. 2002. Long-term survival of direct seeded Wyoming big sagebrush seedlings on a reclaimed mine site. *Arid Land Research and Management* 16:309-317.
- Schuman, G.E., Booth, D.T., Cockrell, J.R. 1998. Cultural methods for establishing Wyoming big sagebrush on mined lands. *Journal of Range Management* 51:223-230.
- Schuman, G.E., Vicklund, L.E., Belden, S.E. 2005. Establishing *Artemisia tridentata* ssp *wyomingensis* on mined lands: science and economics. *Arid Land Research and Management* 19:353-362.
- Sharp, L. 1986. Crested wheatgrass: its values, problems, and myths. *In*: Johnson, K.L. [ED.], *Crested wheatgrass: its values, problems, and myths*, symposium proceeding, Logan, UT: Utah State University. p. 1-34.
- Shaw, N.L., DeBolt, A.M., Rosentreter, R. 2005. Reseeding big sagebrush: techniques and issues. *In*: Shaw, N.L., Pellant, M. and Monsen, S.B. [EDS.]. Sage-grouse habitat restoration. RMRS-P-38., Boise, ID: USDA, Forest Service, Rocky Mountain Research Station. p. 99-108.
- Sika, J.L. 2006. Breeding ecology, survival rates, and causes of mortality of hunted and nonhunted greater sage-grouse in central Montana [Thesis]. Bozeman, Montana State University. 118 p.
- Smoliak, S., Dormaar, J.F. 1985. Productivity of Russian wildrye and crested wheatgrass and their effect on prairie soils. *Journal of Range Management* 38:403-405.
- Stevens, R. 1992. Interseeding and transplanting to enhance species composition. *In*: Monsen, S.B., Kitchen, S.G. [EDS.]. *Ecology and management of annual rangelands*. General Technical Report INT-GTR-313., Ogden, UT: USDA, Forest Service, Intermountain Research Station. p. 300-306.
- Stevens, R. 2004. Establishing plants by transplanting and interseeding. *In*: Monsen, S.B., R. Stevens, R., Shaw, N.L. [EDS.]. *Restoring western ranges and wildlands*. General Technical Report RMRS-GTR-136-vol-3., Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station. p. 739-744.
- Stevens, R., Moden, W.L., McKenzie, D.W. 1981. Interseeding and transplanting shrubs and forbs into grass communities. *Rangelands* 3:55-58.

- Stevens, R., Monsen, S.B. 2004. Guidelines for restoration and rehabilitation of principal plant communities. *In*: Monsen, S.B., Stevens, R., Shaw, N.L. [EDS.]. Restoring western ranges and wildlands. General Technical Report RMRS-GTR-136-vol-1., Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station. p. 199-294.
- Sveum, C.M., Crawford, J.A., Edge, W.D. 1998. Use and selection of brood-rearing habitat by sage grouse in south central Washington. *Great Basin Naturalist* 58:344-351.
- U.S. Fish and Wildlife Service. 2005. Endangered and threatened wildlife and plants, 12-month finding for petitions to list the greater sage-grouse as threatened and endangered, proposed rule. *Federal Register* 70:2244-2282.
- Van Epps, G.A., Mckell, C.M. 1978. Major criteria and procedures for selecting and establishing range shrubs as rehabilitators of disturbed lands. *In*: Hyder, D.N. [ED.], Proc. 1st International Rangeland Congress. 30-31 March 1984, Denver, CO: Society for Range Management. p. 352-354.
- Vickland, L.E., Schuman, G.E., Hild, A.L. 2004. Influence of sagebrush and grass seeding rates on sagebrush density and plant size *In*: Hild, A.L., Shaw, N.L., Meyer, S.E., Booth, D.T., McArthur, E.D. [EDS.]. Seed and Soil Dynamics in Shrubland Ecosystems. 12-16 August 2002, Laramie, WY. Proceedings RMRS-P-31. Ogden, UT: USDA, Forest Service, Rocky Mountain Research Station. p. 40-45.
- Wallestad, R., Peterson, J.G., Eng, R.L. 1975. Foods of adult sage grouse in central Montana. *Journal of Wildlife Management* 39:628-630.
- Wallestad, R., Pyrah, D. 1974. Movement and nesting of sage grouse hens in central Montana. *Journal of Wildlife Management* 38:630-633.
- Wallestad, R.O. 1971. Summer movements and habitat use by sage grouse broods in central Montana. *Journal of Wildlife Management* 35:129-136.
- Wambolt, C.L., Harp, A.J., Welch, B.L., Shaw, N., Connelly, J.W., Reese, K.P., Braun, C.E., Klebenow, D.A., McArthur, E.D., Thompson, J.G., Torell, L.A., Tanaka, J.A. 2002. Conservation of greater sage-grouse on public lands in the western U.S.: Implications of recovery and management policies. PACWPL Policy Paper SG-02-02., Caldwell, Idaho: Policy Analysis Center for Western Public Lands. 41 p.
- Weaver, J.E. 1950. Stabilization of midwestern grassland. *Ecological Monographs* 20:251-270.
- Whisenant, S.G. 1999. Repairing damaged wildlands: a process-orientated, landscape-scale approach. New York: Cambridge University Press. 312 p.
- Williams, M.I., Schuman, G.E., Hild, A.L., Vicklund, L.E. 2002. Wyoming big sagebrush density: effects of seeding rates and grass competition. *Restoration Ecology* 10:385-391.
- Wilson, S.D., Gerry, A.K. 1995. Strategies for mixed-grass prairie restoration: Herbicide, tilling, and nitrogen manipulation. *Restoration Ecology* 3:290-298.
- Wilson, S.D., Partel, M. 2003. Extirpation or coexistence? Management of a persistent introduced grass in a prairie restoration. *Restoration Ecology* 11:410-416.
- Woodward, J.K. 2006. Greater sage-grouse (*Centrocercus urophasianus*) habitat in central Montana [Thesis]. Bozeman, MT: Montana State University. 92 p.
- Young, J.A., Evans, R.A. 1986. History of crested wheatgrass in the Intermountain area. *In*: Johnson, K.L. [ED.], Proc. Crested Wheatgrass: Its Values, Problems, and Myths. 3-7 October 1986, Logan, UT: Utah State University. p. 21-26.
- Young, J.A., Evans, R.A. 1989. Dispersal and germination of big sagebrush (*Artemisia tridentata*) seeds. *Weed Science* 37:201-206.

Clarifying Potential Successional Trajectories in Sagebrush Communities Historically Seeded with Crested Wheatgrass

Kevin L. Gunnell¹, Justin R. Williams², and Thomas A. Monaco²

ABSTRACT

Crested wheatgrass (Agropyron cristatum [L.] Gaertn.) has been historically seeded on thousands of ha of Wyoming big sagebrush (Artemisia tridentata Nutt. ssp. wyomingensis) communities. Initially used to improve degraded rangeland, its use has become controversial in the current management setting where emphasis has shifted from increased forage production to increased community diversification. Naturalized crested wheatgrass communities are an attractive setting for this goal of diversification, but little is understood about how succession proceeds in the absence of management techniques that perpetuate the dominance of crested wheatgrass, in other words, fire, chemical, and mechanical treatments. Further research in communities that have not experienced these treatments may help define interacting factors, determine successional trajectories, and provide insights into management options to satisfy contemporary and future rangeland management activities within seeded Wyoming big sagebrush communities.

INTRODUCTION

Crested wheatgrass (*Agropyron cristatum* [L.] Gaertn.) is an exotic perennial bunchgrass that has been seeded on thousands of ha in the Great Basin. These seeded communities represent one of the largest-scale land management manipulations within sagebrush-steppe ecosystems. Pervasive overgrazing across this ecosystem diminished native herbaceous plant cover and promoted dense stands of Wyoming big sagebrush (*Artemisia tridentata* Nutt. ssp. *wyomingensis*) and the opportunity for weed invasion (Young and others 1999). In the early 1950s, large-scale preventative measures were taken to control the spread of the poisonous weed halogeton (*Halogeton glomeratus* [Bieb.] C.A. Mey.) (Mathews 1986). Federal appropriations were granted to government agencies to prevent further expansion of halogeton and other weeds, improve forage potential, reduce fire frequency, and stabilize soil resources of these degraded landscapes (Blaisdell and others 1982; Mathews 1986). Crested wheatgrass was a primary tool used to accomplish this goal (Young and Evans 1986). Since the 1970s, crested wheatgrass has been planted on private and public lands in the Great Basin primarily following wildfires to maintain rangeland productivity, and reduce the spread of invasive

annual species including downy brome (*Bromus tectorum* L.), Russian thistle (*Salsola kali* L.), tumble mustard (*Sisymbrium altissimum* L.) and medusahead rye (*Taeniatherum caput-medusae* [L.] Nevski). Hundreds of articles were published in scientific journals between 1950 and 1980 on management implications of crested wheatgrass communities. The majority of these articles focused on methodologies to properly manage these communities for forage production while preventing sagebrush from reinvading. In our contemporary era of rangeland management, emphasis has shifted to include stabilization and restoration of shrub-steppe communities. Current rangeland rehabilitation emphases include reducing wildfire frequency, improving species diversity, improving critical winter habitat for big game species, and protecting habitat for threatened and endangered wildlife species (Pellant and Lysne 2005).

Wyoming big sagebrush communities seeded with crested wheatgrass are a particularly attractive setting to improve Great Basin plant communities because they are characteristically more stable and less fire-prone than downy brome-dominated communities. While some studies have evaluated reseeding native species into crested wheatgrass-dominated communities (Bakker and others 1997; Bakker and Wilson 2001; Fansler 2007), studies on the ability of native species to naturally re-establish are lacking. Today, vast landscapes within the Great Basin contain or are dominated by crested wheatgrass. The continued use of crested wheatgrass is controversial because the long-term ecological consequences of its use within this ecosystem are unknown. In addition, while landscapes were stabilized and forage potential was improved by crested wheatgrass, these historic seedings have not been comprehensively characterized to determine their vegetative and seed bank composition, structure, or functional attributes. Specifically, this extensive land-management treatment has not been adequately assessed in terms of the multiple interacting factors that determine vegetation and seed bank attributes and successional trajectories. Filling these gaps in knowledge may provide a mechanistic understanding of the factors limiting natural reinvasion by native species and help overcome the challenges associated with diversifying crested wheatgrass communities. We provide this literature review to address these issues and discuss what is known and directions that further research might take.

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Gunnell, Kevin L., Department of Wildland Resources-Utah State University, Logan, UT

²Justin R. Williams and Thomas A. Monaco, USDA-ARS Forage and Range Research Laboratory, Logan, UT.

Brief History of Great Basin Vegetation

The Intermountain Semi-desert and Desert Province Ecoregions as described by Bailey (1995) contain an area known as the Basin and Range or Great Basin, a high elevation, mountainous, cool desert (Irwin-Williams and others 1990). This distinct land-area consists of many vegetation cover-types (Johnson 1989; Shiflet 1994; Young 1994). In particular, sagebrush-steppe cover types are characterized by a co-dominance of woody shrubs and an herbaceous understory of bunchgrasses and forbs (West 1988). The common native species found today probably occurred in different abundance and composition before historic human-influenced disturbances.

Pre-settlement composition of vegetation is difficult to predict for many reasons. The first explorers in the Great Basin emphasized the dominance of sagebrush. Historical ecology also confirms sagebrush dominance (Rogers 1982; Vale 1975); however, some locations may have been grass-dominated (Hull and Hull 1974). Variability in vegetation dominance in Great Basin sagebrush communities has been attributed to dynamic variation in fire frequency, climate, and herbivory (Egan and Howell 2001; Knight 1994; Mack and Thompson 1982; Swetnam and others 1999; West and Yorks 2006; Wright 1985).

Considerable evidence exists indicating that regardless of sagebrush dominance, native grass and forb abundance has drastically declined (Johnson 1989; West 1988). Since large herbivores were not present in numbers to adversely impact herbaceous species, the decline in abundance has been attributed to poor grazing tolerance and the absence of intensive ungulate herbivory prior to European settlement (Mack and Thompson 1982). Wildfires did occur, and because herbaceous species have the potential to readily regrow while sagebrush does not (Wright 1985), wildfire functioned as an ecosystem driver to determine relative abundances of sagebrush and herbaceous cover (Bork and others 1998; Harniss and Murray 1973; Kay 1960; Monsen 1994; Wright 1985).

Fire frequency within sagebrush-steppe ecosystems has also changed considerably since European settlement. Changes in fire frequency were initiated as a consequence of invasive annual grasses replacing native herbaceous species (Monsen 1994; Young and Evans 1973). Foremost among these annual grasses are downy brome and medusahead (*Taeniatherum caput-medusae* (L.) Nevski), which produce high amounts of fine-fuels that carry fire more readily than native species (Whisenant 1990; Young and Evans 1973). The spread and dominance of annual weeds produced larger wildfires and perpetuated further expansion of annual weed infestations. Introduction of livestock grazing and annual weed-mediated changes in wildfire frequency greatly reduced the forage potential of sagebrush-steppe ecosystems (Griffiths 1903; Pickford 1932; Tournay 1894).

Consequently, land management efforts attempted to end the misuse of sagebrush-steppe rangelands in the early 1900s (Stoddart and Smith 1955); however, overgrazing and wildfire continued to alter this ecosystem well into the twentieth century because of the demand for livestock production (Chatterton and Young 2002; Young 1994). In this era, improving the forage potential of these lands was the primary goal of resource managers (Young and Evans 1986).

Introduction of Crested Wheatgrass Into The Great Basin

Identifying forage grasses to improve rangeland productivity was the research emphasis of early American range researchers such as P.B. Kennedy and A.W. Sampson at the turn of the 20th century (Chatterton and Young 2002). Many exotic grasses, including crested wheatgrass performed better than the available native species, and became the “gold standard” to use on former shrub-steppe and sagebrush rangelands (Young and McKenzie 1982). Crested wheatgrass was first introduced into the Northern Great Plains from Russia in 1898 and was later promoted as a successful revegetation species suitable for the needs of land managers (Dillman 1946; Rogler and Lorenz 1983; Young and Evans 1986). The first seeding evaluations in the northern Great Basin and Snake River Plain were made in the early 1930s (Hull and Klomp 1966). Reclamation of abandoned dry-land farms and unproductive sagebrush pastures on private and public land prompted government researchers to publish bulletins on how to reseed these rangelands in Nevada, Idaho, and Utah in the early 1940s (Pellant and Lysne 2005; Young and Evans 1986). Less productive sagebrush-steppe grazing areas were plowed and seeded with crested wheatgrass to increase forage production, promote the rehabilitation of degraded rangelands, and increase meat production to support the World War II effort (Young and Evans 1986). Eventually, the use of crested wheatgrass greatly expanded in 1952 with the Halogeton Control Act, a program designed to stop the spread of the poisonous, annual weed halogeton (*Halogeton glomeratus* [M. Bieb.] C.A. Mey.) by seeding crested wheatgrass throughout the Great Basin and beyond (Young and Evans 1986).

Establishing crested wheatgrass required modifying traditional agricultural techniques to fit a new set of obstacles associated with seedings on rough rangelands. Developing the rangeland drill and brushland plow effectively allowed sagebrush removal and large seedings over rough terrain (Young and McKenzie 1982). Large-scale seedings continued until the 1970s, when crested wheatgrass became the focus of controversy (Chatterton and Young 2002; Pellant and Lysne 2005). This controversy stemmed from the perception that crested wheatgrass may be responsible for low quality wildlife habitat and diminished native plant diversity (Dobkin and

Sauder 2004; Wilson 1989). Thus, federal agencies were encouraged to use native species in new seeding projects when feasible depending on availability, cost, and potential success for establishment (Pellant and others 2004; Pellant and Lysne 2005). Crested wheatgrass is still used in large-scale seedings on public land, but is now a component of a more diverse seed mix that includes native species (Pellant and Lysne 2005). The underlying issue today is that crested wheatgrass exists on millions of has of public and private rangelands and is a persistent, naturalized component of plant communities where it was established in the past.

As the need to reclaim degraded rangelands increased, the production of crested wheatgrass seed also increased (Sharp 1986). Thus, in addition to being easier to establish than native grasses, plentiful seed availability also led to its preference for rangeland seedings. Good seedling vigor makes crested wheatgrass ideal for reclamation because it is very competitive with slower growing species, is drought tolerant, and has an efficient nutrient acquiring root system (Bakker and Wilson 2001; Caldwell and others 1991; Eissenstat and Caldwell 1988). In addition, crested wheatgrass is competitive with weedy species and reduces downy brome dominance better than native species because it germinates earlier and grows more rapidly at colder temperatures (Chatterton and Harrison 2003). Crested wheatgrass is long-lived and perpetuates stable populations primarily by good seed production (Marlette and Anderson 1986). Forage production is two-fold greater than native range in shrub-steppe ecosystems with limited precipitation (Angell and others 1990; Gade and Provenza 1986; Ganskopp and others 1997; Laycock and Conrad 1981; Springfield and others 1967). Crested wheatgrass is also known to have superior drought tolerance (Caldwell and Richards 1986; Sharp and others 1992) and can persist under drought conditions in semi-arid environments (Busso and others 1990; Currie and Peterson 1966). In general, crested wheatgrass can withstand heavy grazing (Angell 1997; Caldwell and others 1991; Cook and others 1958; Laycock and Conrad 1981; Sharp 1986) with some limitations (Sharp 1970). Because crested wheatgrass evolved with ungulate grazing pressure, it is well adapted to higher utilization levels than native Great Basin grasses (Sharp 1986). Crested wheatgrass can withstand intense biomass removal because of its ability to re-establish pre-defoliation leaf area and high compensatory photosynthesis (Frischknecht and Harris 1968; Olsen and Richards 1988).

Succession in Sagebrush Communities Seeded With Crested Wheatgrass

Early research on crested wheatgrass management focused on grazing and fire as well as other mechanical and chemical treatments to maintain crested wheatgrass productivity and dominance (Evans and Young 1978; Hull 1974; Klomp and Hull 1972; McLean and van Ryswyk 1973; Mueggler and Blaisdell 1958). Although crested

wheatgrass reduces shrub seedling invasion more than native grasses (Blaisdell 1949; Frischnecht and Bleak 1957; Schuman and others 1982), without fire and subsequent management treatments, sagebrush reinvades over time (Frischknecht and Bleak 1957; Hull and Klomp 1966). A review of these management activities is a necessary and preliminary step to determine the potential of seedings to establish as stable crested wheatgrass stands or be reinvaded by native sagebrush-steppe species from adjacent plant communities (Johnson and Payne 1968).

Livestock grazing practices, including intensity, duration, season, and animal type impact plant succession in sagebrush communities seeded with crested wheatgrass (Holechek and others 1995; Pieper 1994). Crested wheatgrass dominance is maintained, and sagebrush encroachment is reduced, by heavy autumn grazing more so than spring grazing (Frischknecht and Harris 1968; Hull and Klomp 1974; Laycock and Conrad 1981; Robertson and others 1970). In addition, sheep versus cattle grazing of crested wheatgrass reduces shrub encroachment (Blaisdell and others 1982; Bleak and Plummer 1954; Laycock and Conrad 1981). Livestock grazing can also directly reduce shrub invasion by trampling seedlings (Owens and Norton 1992). Conversely, livestock grazing practices can also promote sagebrush reinvasion, particularly prolonged spring grazing during drought (Busso and Richards 1995). Over-utilization of crested wheatgrass through high intensity, long-duration grazing reduces grass productivity and enhances seedling survival of other species including sagebrush (Angell 1997; Salihi and Norton 1987). Grazing crested wheatgrass in the spring reduces its dominance and may allow invasion of shrub species (Laycock and Conrad 1981; Olsen and Richards 1988; Robertson and others 1970). However, regardless of grazing, and without periodic treatments to reduce sagebrush, it is capable of reinvading crested wheatgrass seedings (Astroth and Frischknecht 1984; Torell 1986).

Management activities that directly kill shrub and herbaceous dicotyledonous species have obvious impacts on maintaining crested wheatgrass dominance. The absence of fire and brush removal treatments (chemical and mechanical) allows crested wheatgrass seedings to be reinvaded by native sagebrush-steppe species from adjacent plant communities. Prescribed fire perpetuates crested wheatgrass dominance because it has a high post-burn recovery and productivity, whereas sagebrush is killed (Harniss and Murray 1973; Kay 1960; Lodge 1960; Wright 1985). Mechanical treatments, including plowing, disking, chaining, mowing, and the use of pipe harrows, rails, and cables effectively reduce shrub abundance and density, and facilitate herbaceous dominance (Blaisdell and others 1982; Lodge 1960; Mueggler and Blaisdell 1958; Parker 1979; Pechanec and others 1965; Vallentine 1989; Wambolt and Payne 1986; Young and McKenzie 1982). Finally, chemical

control with 2, 4-D and tebuthiuron successfully reduces shrub density and increases crested wheatgrass dominance (Astroth and Frischknecht 1984; Cox and Anderson 2004; Eckert and others 1972; Evans and others 1979; Holechek and others 1995; Johnson and Payne 1968; Pellant and Lysne 2005). The effectiveness of these treatments to maintain crested wheatgrass dominance is also determined by their season of application (Blaisdell and others 1982; Evans and Young 1978).

Less obvious non-management factors may also influence successional trajectories in sagebrush communities seeded with crested wheatgrass. First, the resilience and dominance of crested wheatgrass seedlings depends on how well the initial seeding establishes (Hull 1974). Variability in seedbed preparation, including weed and shrub control, determines soil moisture retention and the competitive ability of crested wheatgrass seedlings (Holechek and others 1995; Hull 1974; Klomp and Hull 1972; McLean and van Ryswyk 1973; Mueggler and Blaisdell 1958; Vallentine 1989). Second, soil properties may be a good predictor of successional trajectories because soil nutrients, aggregate stability, bulk density, soil penetration resistance, and water infiltration are known to remain stable under long-term livestock grazing in crested wheatgrass seedlings compared to adjacent native-dominated rangeland (Krzic and others 2000). Third, variability in seasonal precipitation and topography are not only the most influential factors determining vegetation dynamics in sagebrush communities (Cook and Irwin 1992; West and Yorks 2006), but may also interact with management activities to determine vegetation status of seeded sagebrush communities (Sharp and others 1992; West 1988). However, management activities that remove woody and herbaceous dicotyledonous species obscure the role that these indirect factors play in determining secondary succession within seeded communities because of the obvious impact shrub removal has had on maintaining crested wheatgrass dominance.

Seed Bank Dynamics of Sagebrush Communities Seeded With Crested Wheatgrass

Seed banks provide important insights into plant community dynamics including colonization and succession (Bazzaz 1979), population structure (Wilson 1992), and community structure (Yarranton and Morrison 1974). Seed banks are characterized by their horizontal and vertical distribution within the soil and litter, and temporally according to the timing of seed rain input and seed longevity (Freas and Kemp 1983; Kemp 1989; Reichman 1975; Williams and others 1974; Young and Evans 1975). Transient seed banks are viable for less than 1 year and therefore rely on seed rain input to maintain a presence in the seed bank, while persistent seed banks are viable longer than 1 year and can persist in the seed bank without further seed rain input (Thompson and Grime 1979). In general, seed banks of annual species are more persistent than

perennial species in arid and semiarid ecosystems (Kemp 1989). No studies were found that addressed how the seed banks of sagebrush communities seeded with crested wheatgrass were horizontally or vertically distributed, but the temporal distribution of crested wheatgrass was less variable than the native bluebunch wheatgrass (Pyke 1990). While bluebunch wheatgrass released all of its seed upon maturation, crested wheatgrass retained seed in the inflorescence and released them into the following spring. This helped crested wheatgrass to maintain a viable seed bank throughout the growing season and to protect its seed from predation (Pyke 1990). It was also observed that ungerminated crested wheatgrass seeds stayed viable in the seed bank from the first to second year of the study (Pyke 1990).

There is still little known about seed bank composition and its impact on succession within Wyoming big sagebrush communities seeded with crested wheatgrass over a broad representative area. Composition of seed banks in semi-arid ecosystems seeded with crested wheatgrass generally resembles the current aboveground vegetation (Ambrose and Wilson 2003; Henderson and others 1988; Henderson and Naeth 2005; Marlette and Anderson 1986; Pyke 1990). The high seed productivity of crested wheatgrass (Pyke 1990) and the transient nature of the seed bank of many native species in these communities can create a negative feedback that perpetuates crested dominance in the seed bank. The density of crested wheatgrass seeds in the seed bank of sagebrush communities seeded with crested wheatgrass was typically much higher than for other species. Pyke (1990) observed densities of 1,772 to 1,037 seeds m^{-2} and 26 to 0 seeds m^{-2} for crested wheatgrass and bluebunch wheatgrass, respectively, for two years of the study (1986 to 1987). Marlette and Anderson (1986) showed similar dominance of crested wheatgrass in the seed bank, with 84.67 to 92.28 percent of low diversity stands and 31.91 to 72.79 percent in sagebrush re-invaded communities. Crested wheatgrass may also dominate seed banks of these communities because of the lack of a seed source and the poor dispersal of native species (Marlette and Anderson 1986). Seed bank composition, or more specifically, seed density of species, may exert some control over successional trajectories of plant communities through its effects on interference between emerging seedlings (Francis and Pyke 1996; Sheley and Larson 1994; Velagala and others 1997). When crested wheatgrass dominates seed banks, it may have a disproportionate impact on seedling emergence and establishment of native species because of the well-recognized advantages for seedling performance of crested wheatgrass (Bakker and Wilson 2001; Caldwell and others 1991; Eissenstat and Caldwell 1988; Harris and Wilson 1970).

Negative feedbacks that maintain crested wheatgrass dominance in the seed bank can be minimized if treatments to eradicate sagebrush and other shrubs are ineffective (Johnson and Payne 1968; Marlette and Anderson 1986). Higher shrub cover creates areas of increased seed capture (Guo and others 1998; Nelson and Chew 1977), increased resource availability, and seed safe sites (Hassan and West 1986; Pugnaire and Lazaro 2000; Young and Evans 1975). Preferential livestock use of crested wheatgrass over most shrub species creates an opportunity to use livestock to manipulate the seed banks of crested wheatgrass communities. Dominance of crested wheatgrass in the seed bank can be reduced by repetitive heavy spring grazing since grazing directly impacts crested wheatgrass seed production and competition and creates opportunities for sagebrush to establish (Frischknecht and Bleak 1957; Harris and others 1968; Hull and Klomp 1966; Rittenhouse and Sneva 1976).

IMPLICATIONS

Crested wheatgrass was initially adopted to restore the productivity of depleted rangelands. However, there are concerns about low diversity and the long-term effects introduced plants have on native landscapes, in particular, wildlife habitat, soil quality, and ability to resist invasive weeds (Dobkin and Sauder 2004; Dormaar and others 1995; Holechek 1981; Wilson 1989). Many of these concerns are based on the lack of research in historic (> 30 year-old) crested wheatgrass seedings. Therefore, addressing important issues involving these seeded communities may help managers make important decisions for future management. Do seeded Wyoming big sagebrush communities remain dominated by crested wheatgrass and resistant to weed invasion? What is the variability in species abundance and diversity in old crested wheatgrass seedings, and are they functionally different than native sagebrush-steppe rangelands managed similarly? Is the perception of low plant diversity in crested wheatgrass seedings an artifact perpetuated by obvious management activities and pre-seeding soil attributes that favor crested wheatgrass dominance? Is resilience of crested wheatgrass caused by seed limitation of native species or competitive exclusion of seeds that arrive? What is the role of grazing on plant species diversity, relative to the factors of soil and climate? These issues can be addressed through the practical approach of evaluating the relationships between multiple interacting factors, in other words,, vegetation, grazing history, soil properties, climate, topography, and seed banks on Wyoming big sagebrush communities historically seeded with crested wheatgrass that have not experienced fire or other mechanical and chemical treatments. Further research to address these questions may help define how these multiple interacting factors determine successional trajectories and provide insights into

management options to satisfy contemporary and future rangeland management activities within seeded Wyoming big sagebrush communities.

REFERENCES

- Ambrose, L.G., Wilson, S.D. 2003. Emergence of the Introduced Grass *Agropyron cristatum* and the Native Grass *Bouteloua gracilis* in a Mixed-grass Prairie Restoration. *Restoration Ecology* 11:110-115.
- Angell, R.A. 1997. Crested wheatgrass and shrub response to continuous or rotational grazing. *Journal of Range Management* 50:160-164.
- Angell, R.F., Miller, R.F., Haferkamp, M.R. 1990. Variability of crude protein in crested wheatgrass at defined stages of phenology. *Journal of Range Management* 43:186-189.
- Astroth, K.A., Frischknecht, N.C. 1984. Managing Intermountain Rangelands: Research on the Benmore Experimental Range, 1940-84. General Technical Report INT-GTR-175. Ogden, UT, USDA, Forest Service. 44 p.
- Bailey, R.G. 1995. Description of the ecoregions of the United States. 2nd edition. Miscellaneous Publication Number 1391. Washington DC, USDA, Forest Service. 85 p.
- Bakker, J.D., Christian, J., Wilson, S.D., Waddington, J. 1997. Seeding blue grama in old crested wheatgrass fields in southwest Saskatchewan. *Journal of Range Management* 50:156-159.
- Bakker, J., Wilson, S. 2001. Competitive abilities of introduced and native grasses. *Plant Ecology* 157:117-125.
- Bazzaz, F.A. 1979. The physiological ecology of plant succession. *Annual Review of Ecology and Systematics* 10:351-371.
- Blaissdell, J.P. 1949. Competition between sagebrush seedlings and reseeded grasses. *Ecology* 30 512-519.
- Blaissdell, J.P., Murray, R.B., McArthur, E.D. 1982. Managing intermountain rangelands-sagebrush-grass ranges. Report INT-GTR-134. Ogden, UT: Intermountain Forest and Range Experiment Station. 41 p.
- Bleak, A.T., Plummer, A.P. 1954. Grazing crested wheatgrass by sheep. *Journal of Range Management* 7:63-68.
- Bork, E.W., West, N.E., Walker, J.W. 1998. Cover components on long-term seasonal sheep grazing treatments in three-tip sagebrush steppe. *Journal of Range Management* 51:293-300.
- Busso, C.A., Richards, J.H. 1995. Drought and clipping effects on tiller demography and growth of two tussock grasses in Utah. *Journal of Arid Environments* 29:239-251.
- Busso, C.A., Richards, J.H., Chatterton, N.J. 1990. Nonstructural carbohydrates and spring regrowth of two cool-season grasses: interaction of drought and clipping. *Journal of Range Management* 43:336-343.
- Caldwell, M.M., Manwaring, J.H., Jackson, R.B. 1991. Exploitation of phosphate from fertile soil micro-sites by 3 Great Basin perennials when in competition. *Functional Ecology* 5:757-764.
- Caldwell, M.M., Richards, J.H. 1986. Competing root systems: morphology and models of absorption. In: T.J. Givnish [ed.]. 1989. On the economy of plant form and function. New York, NY, USA: Cambridge University Press. p. 251-273.
- Chatterton, N.J., Young, J.A. 2002. Grasses in the West: Early Days to Present. *Journal of the West* 41:25-35.
- Chatterton, N.J., Harrison, P.A. 2003. Fructans in crested wheat grass leaves. *Journal of Plant Physiology* 160:843-849.
- Cook, J.G., Irwin, L.L. 1992. Climate-vegetation Relationships between the Great Plains/ Great Basin. *American Midland Naturalist* 127:316-326.
- Cook, C.W., Stoddart, L.A., Kinsinger, F.E. 1958. Responses of crested wheatgrass to various clipping treatments. *Ecological Monographs* 28:237-273.
- Cox, R.D., Anderson, V.J. 2004. Increasing native diversity of cheatgrass-dominated rangeland through assisted succession. *Journal of Range Management* 57:203-210.
- Currie, P.O., Peterson, G. 1966. Using growing-season precipitation to predict crested wheatgrass yield. *Journal of Range Management* 19:284-288.
- Dillman, A.C. 1946. The beginnings of crested wheatgrass in North America. *Journal American Society Agronomy* 38:237-250. Dobkin,

- D.S., Sauder J.D. 2004. Shrubsteppe landscapes in jeopardy. Distributions, abundances, and the uncertain future of birds and small mammals in the Intermountain West. Bend, OR, USA: High Desert Ecological Research Institute. 47 p.
- Dormaar, J.F., Naeth, M.A., Willms, W.D., Chanasyk, D.S. 1995. Effect of native prairie, crested wheatgrass and Russian wildrye on soil chemical properties. *Journal of Range Management* 48:258-263.
- Eckert, R.E., Jr., Bruner, A.D., Klomp, G.I. 1972. Response of understory species following herbicidal control of low sagebrush. *Journal of Range Management* 25: 280-285.
- Egan, D., Howell, E.A. 2001. The historical ecology handbook. A restorationist's guide to reference ecosystems. Washington D.C., USA: Island Press. 484 p.
- Eissenstat, D.M., Caldwell, M.M. 1988. Competitive ability is linked to rates of water extraction: a field study of two aridland tussock grasses. *Oecologia* 75:1-7.
- Evans, R.A., Young, J.A. 1978. Effectiveness of rehabilitation practices following wildfire in a degraded big sagebrush-downy brome community. *Journal of Range Management* 31:185-188.
- Evans, R.A., Young, J.A., Eckert, R.E., Jr. 1979. Use of herbicides as a management tool. In: The sagebrush ecosystem: Symposium. 1978. Logan, College of Natural Resources, USU p. 110-116.
- Fansler, V.A. 2007. Establishing native plants in crested wheatgrass stands using successional management [thesis]. Corvallis, OR, USA: Oregon State University. 108 p.
- Francis, M.G., Pyke, D.A. 1996. Crested wheatgrass-cheatgrass seedling competition in a mixed-density design. *Journal of Range Management* 49:432-438.
- Freas, K.E., Kemp, P.R. 1983. Some relationships between environmental reliability and seed dormancy in desert annual plants. *Journal of Ecology* 72:211-217.
- Frischknecht, N.C., Bleak, A.T. 1957. Encroachment of big sage brush on seeded range in northeastern Nevada. *Journal of Range Management* 10:165-170.
- Frischknecht, N.C., Harris, L.E. 1968. Grazing intensities and systems on crested wheatgrass in central Utah: response of vegetation and cattle. Technical Bulletin 1388. Washington, D.C., USA: USDA. 47 p.
- Gade, A.E., Provenza, F.D. 1986. Nutrition of sheep grazing crested wheatgrass versus crested wheatgrass-shrub pastures during winter. *Journal of Range Management* 39:527-530.
- Ganskopp, D., Myers, B., Lambert, S., Cruz, R. 1997. Preferences and behavior of cattle grazing 8 varieties of grasses. *Journal of Range Management* 50:578-586.
- Guo, Q., Rundel, P.W., Goodall, D.W. 1998. Horizontal and vertical distribution of desert seed banks: patterns, causes, and implications. *Journal of Arid Environments* 38:465-478.
- Griffiths, D. 1903. Forage conditions and problems in eastern Washington, eastern Oregon, north-eastern California, and north-western Nevada. Bulletin No. 38. Washington D.C., USDA, Bureau of Plant Industry. p. 5.
- Harniss, R.O., Murray, R.B. 1973. 30 years of vegetal change following burning sagebrush-grass range. *Journal of Range Management* 26:322-325.
- Harris, L.E., Frischknecht, N.C., Sudweeks, E.M. 1968. Seasonal grazing of crested wheatgrass by cattle. *Journal of Range Management* 21:221-225.
- Harris, G.A., Wilson, A.M. 1970. Competition for moisture among seedlings of annual and perennial grasses as influenced by root elongation at low temperature. *Ecology* 51:530-534.
- Hassan, M.A., West, N.E. 1986. Dynamics of soil seeds pools in burned and unburned sagebrush deserts. *Ecology* 67:269-272.
- Henderson, D.C., Naeth, M.A. 2005. Multi-scale impacts of crested wheatgrass invasion in mix-grass prairie. *Biological Invasions* 7:639-650.
- Henderson, C.B., Petersen, K.E., Redak, R.A. 1988. Spatial and temporal patterns in the seed bank and vegetation of a desert grassland community. *The Journal of Ecology* 76:717-728.
- Holecck, J.L. 1981. Brush control impacts on rangeland wildlife. *Journal of Soil and Water Conservation* 36:265-269.
- Holecck, J.L., Pieper, R.D., Herbal, C.H. 1995. Range management: principles and practices. 2nd edition. Upper Saddle River, NJ, USA: Prentice Hall. 526 p.
- Hull A. C., Jr. 1974. Species for seeding arid rangeland in southern Idaho. *Journal of Range Management* 27:216-218.
- Hull A.C., Jr. 1974. Pre-settlement vegetation of Cache Valley Utah and Idaho. *Journal of Range Management* 27:27-29.
- Hull, A.C., Jr., Klomp, G.J. 1966. Longevity of crested wheatgrass in the sagebrush-grass type in Idaho. *Journal of Range Management* 19: 5-11.
- Hull, A.C., Jr., Klomp, G.J. 1974. Yield of crested wheatgrass under four densities of big sagebrush in Southern Idaho. Technical Bulletin 1483. Washington, D.C., USA: USDA, Agricultural Research Service. 38 p.
- Irwin-Williams, C.C., et al. 1990. Man and plants in the Great Basin. In: Osmond, C.B. [eds.]. Plant biology of the Basin and Range. Ecological Studies, Vol. 80. Springer-Verlag. 375 p.
- Johnson, K.L., [ed.]. 1989. Rangeland Resources of Utah. Coop-Extension Service, Utah State University. Logan, UT. 103 p.
- Johnson, J.R. 1968. Sagebrush reinvasion affected by some environmental influences. *Journal of Range Management* 21:209-213.
- Kay, B.L. 1960. Effect of fire on seeded forage species. *Journal of Range Management* 13: 31-33.
- Kemp, P.R. 1989. Seed banks and vegetation processes in deserts. In: Leck, M.A., et al. [eds.]. 1989. Ecology of soil seed banks. San Diego, CA, USA: Academic Press Limited. p. 257-281.
- Klomp, G.J., Hull, A.C., Jr. 1972. Methods for seeding three perennial wheatgrasses on cheatgrass ranges in southern Idaho. *Journal of Range Management* 25:266-268.
- Knight, D.H. 1994. Mountains and Plains: The ecology of Wyoming landscapes. New Haven and London, United Kingdom: Yale University Press. 338 p.
- Krzic, M., Broersma, K., Thompson, D.J., Bomke, A.A. 2000. Soil properties and species diversity of grazed crested wheatgrass and native rangelands. *Journal of Range Management* 53:353-358.
- Laycock, W.A., Conrad, P.W. 1981. Responses of vegetation and cattle to systems of grazing on seeded and native mountain range lands in Eastern Utah. *Journal of Range Management* 34:52-58.
- Lodge, R.W. 1960. Effects of burning, cultivating, and mowing on the yield and consumption of crested wheatgrass. *Journal of Range Management* 13:318-321.
- Mack, R.N., Thompson, J.N. 1982. Evolution in steppe with few large, hooved mammals. *The American Naturalist* 119:757-773.
- Marlette, G.M., Anderson, J.E. 1986. Seed banks and propagule dispersal in crested-wheatgrass stands. *Journal of Applied Ecology* 23: 161-175.
- Mathews, W.L. 1986. Early use of crested wheatgrass seedlings in halogeton control. In: Johnson, K.L. [ed.]. Crested wheatgrass: symposium proceedings, 3-7 October 1983, Utah State University, Logan. p. 27-28.
- McLean, A., van Ryswyk, A.L. 1973. Mortality in crested wheat grass and wildrye. *Journal of Range Management* 26:431-433.
- Monsen, S.B. 1994. Selection of plants for fire suppression on semiarid sites. In: Monsen, S.B. [ed.]. Proceedings-ecology and management of annual rangelands, 18-22 May 1992, Boise, ID. General Technical Report INT-GTR-313. Ogden, UT, Intermountain Research Station. p. 363-373.
- Mueggler W.F., Blaisdell, J.P. 1958. Effects on associated species of burning, rotobating, spraying, and railing sagebrush. *Journal of Range Management* 11:61-66.
- Nelson, J.F., Chew, R.M. 1977. Factors affecting seed reserves in the soil of a Mojave desert ecosystem, Rock Valley, Nye county, Nevada. *American Midland Naturalist* 97:300-320.
- Olsen, B.E., Richards, J.H. 1988. Annual replacement of the tillers of *Agropyron desertorum* following grazing. *Oecologia* 76:1-6.
- Owens, M.K., Norton, B.E. 1992. Interactions of grazing and plant protection on basin big sagebrush seedling survival. *Journal of Range Management* 45:257-262.
- Parker, K.G. 1979. Use of mechanical methods as a management tool. In: The sagebrush ecosystem: symposium. 1978. Logan, College of Natural Resources, Utah State University. p. 117-120.
- Pechanec, J.R., Plummer, P.A., Robertson, J.H., Hull, A.C. 1965. Sagebrush control on rangelands. Handbook 277. USDA. 40 p.
- Pellant, M., Abbey, B., Karl, S. 2004. Restoring the Great Basin desert, U.S.A.: integrating science, management, and people. *Environmental Monitoring and Assessment* 99:169-179.
- Pellant, M., Lysne, C.R. 2005. Strategies to enhance plant structure and diversity in crested wheatgrass seedlings. Proceedings RMRS-P-38. USDA, Forest Service. p. 81-92.
- Pieper, R.D. 1994. Ecological implications of livestock grazing. In: Vavra, M. and others [eds.]. Ecological implications of livestock grazing in the West. Denver, CO, USA: Society of Range Management. p. 177-211.
- Pickford, G.D. 1932. The influence of continued heavy grazing and of promiscuous burning on spring-fall ranges in Utah. *Ecology* 13:159-171.
- Pugnaire, F.I., Lazaro, R. 2000. Seed bank understorey species composition in a semi-arid environment: the effect of shrub age and rainfall. *Annals of Botany* 86:807-813.

- Pyke, D.A. 1990. Comparative demography of co-occurring introduced and native tussock grasses: persistence and potential expansion. *Oecologia* 82:537-543.
- Reichman, O.J. 1975. Relation of desert rodent diets to available resources. *Journal of Mammology* 56:731-749.
- Rittenhouse, L.R., Sneva, F.A. 1976. Expressing the competitive relationship between Wyoming big sagebrush and crested wheatgrass. *Journal of Range Management* 29:326-327.
- Robertson, J.H., Neal, D.L., McAdams, K.R., Tueller, P.T. 1970. Changes in crested wheatgrass ranges under different grazing treatments. *Journal of Range Management* 23:27-34.
- Rogers, G.F. 1982. Then and now: A photographic history of vegetation change in the central Great Basin Desert. Salt Lake City, UT, USA: University of Utah Press. 152 p.
- Rogler, G.A., Lorenz, R.J. 1983. Crested wheatgrass-early history in the United States. *Journal of Range Management* 36:91-93.
- Salihi, D.O., Norton, B.E. 1987. Survival of perennial grass seedlings under intensive grazing in semi-arid rangelands. *Journal of Applied Ecology* 24:145-151.
- Schuman, G.E., Rauzi, F., Booth, D.T. 1982. Production and competition of crested wheatgrass-native grass mixtures. *Agronomy Journal* 74:23-26.
- Sharp, L.A. 1970. Suggested management programs for grazing crested wheatgrass. Bulletin No. 4. Moscow, University of Idaho, Forest, Wildlife and Range Experiment Station. 19 p.
- Sharp, L.A. 1986. Crested wheatgrass: its values, problems and myths. *In: Johnson, K.L. [ed.]. Crested wheatgrass: its values, problems, and myths. Proceedings, 1983 Oct. 3-7. Logan, UT, Utah State University. p. 3-6.*
- Sharp, L.A., Sanders, K., Rimbey, N. 1992. Variability of crested wheatgrass production over 35-years. *Rangelands* 14:153-168.
- Sheley, R.L., Larson, L.L. 1994. Comparative growth and interference between cheatgrass and yellow starthistle seedlings. *Journal of Range Management* 47:470-474.
- Shiflet, T.N. 1994. Rangeland cover types of the United States. Denver, CO, USA: Society for Range Management. 152 p.
- Springfield, H.W., Reid, E.H. 1967. Crested wheatgrass for spring grazing in northern New Mexico. *Journal of Range Management* 20:406-408.
- Stoddart, L.A., Smith, A.D. 1955. *Range Management*. 2nd edition. McGraw-Hill. 433 p.
- Swetnam, T.W., Allen, C.D., Betancourt, J.L. 1999. Applied historical ecology: using the past to manage for the future. *Ecological Applications* 9:1189-1206.
- Thompson K., Grime, J.P. 1979. Seasonal variation in the seed banks of herbaceous species in ten contrasting habitats. *The Journal of Ecology* 67:893-921.
- Torell, L.A. 1986. Economic value of crested wheatgrass: A case study. *In: Johnson, Kendall L. [ed.]. Crested wheatgrass: its values, problems and myths. Proceedings, 1983 Oct. 3-7. Logan, UT, USA: Utah State University. p. 305-310.*
- Tourney, I.W. 1894. The gradual disappearance of the range grasses of the west. *Science* 23:9.
- Vale, T.R. 1975. Presettlement vegetation in the sagebrush-grass area of the Intermountain West. *Journal of Range Management* 28:32-36.
- Vallentine, J.F. 1989. *Range development and improvements*. 3rd edition. San Diego, CA, USA: Academic Press, Inc. 524 p.
- Velagala, R.P., Sheley, R.L., Jacobs, J.S. 1997. Influence of density on intermediate wheatgrass and spotted knapweed interference. *Journal of Range Management* 50:523-529.
- Wambolt C.L., Payne, G.F. 1986. An 18-year comparison of control methods for Wyoming big sagebrush in southwestern Montana. *Journal of Range Management* 39:314-319.
- West, N.E., 1988. Intermountain deserts, shrub steppes, and wood lands. *In: Barbour, M.G., Billings, W.D. [eds.]. North American terrestrial vegetation. Cambridge University Press. p. 209-230.*
- West, N.E., Yorks, T.P. 2006. Long-term interactions of climate, productivity, species richness, and growth form in relictual sagebrush steppe plant communities. *Western North American Naturalist* 66:502-526.
- Whisenant, S.G. 1990. Changing fire frequencies on Idaho's Snake River Plains: Ecological and management implications. *In: McArthur, E.D., et.al. [eds.]. Symposium on cheatgrass invasion, shrub die-off, and other aspects of shrub biology and management. General Technical Report INT-276. Ogden, UT, Forest Service, Intermountain Research Station. p. 4-10.*
- Williams, W.A., Cook, O.D., Kay, B.L. 1974. Germination of native desert shrubs. *California Agriculture* 28:13.
- Wilson, M.F. 1992. The ecology of seed dispersal. *In: Fenner, M. [ed.]. 1992. Seeds: The ecology of regeneration in plant communities. Wallingford, UK: CAB International. p. 61-85.*
- Wilson, S.D. 1989. The suppression of native prairie by alien species introduced for revegetation. *Landscape Urban Planning* 17:113-119.
- Wright, H.A. 1985. Effects of fire on grasses and forbs in sage brush-grass communities. *In: Sanders, K. [ed.]. Rangeland fire effects. Proceedings, Nov. 27-29, 1984. Boise: ID, USDI, Bureau of Land Management. 124 p.*
- Yarranton, G.A., Morrison, R.G. 1974. Spatial dynamics of a primary succession: nucleation. *Journal of Ecology* 62:417-428.
- Young, J.A. 1994. History and use of semiarid plant communities-changes in vegetation. *In: Monsen, S.B., Kitchen, S.G. [comps.]. Proceedings: ecology and management of annual rangelands, 18-21 May 1992, Boise, ID. General Technical Report INT-GRT-313. Ogden, UT, USDA, Intermountain Research Station. p. 5-8.*
- Young, J.A., Evans, R.A. 1973. Downy brome: intruder in the plant succession of big sagebrush communities in the Great Basin. *Journal of Range Management* 26:410-415.
- Young, J.A., Evans, R.A. 1975. Germinability of seed reserves in a big sagebrush community. *Weed Science* 23:358-364.
- Young, J.A., Evans, R.A. 1986. History of crested wheatgrass in the Intermountain area. *In: Johnson, K.L. [ed.]. 1986. Crested wheatgrass: its values, problems and myths. Proceedings, 1983 Oct. 3-7. Logan, UT, USA: Utah State University. p. 21-25.*
- Young J.A., Martinelli, R.A., Eckert, R.E., Jr., Evans, R. A. 1999. Halogeton: A history of mid-20th century range conservation in the Intermountain area. Miscellaneous Publication Number 1553. Reno, NV, USA: USDA, Agricultural Research Service. 60 p.
- Young, J.A., McKenzie, D. 1982. Rangeland drill. *Rangelands* 4:108-113.

Effect of Atmospheric CO₂ Levels on Nutrients in Cheatgrass Tissue

Robert R. Blank¹, Tye Morgan¹, Lewis H. Ziska², and Robert H. White³

ABSTRACT

*Rising atmospheric CO₂ has resulted in declining tissue nutrient concentrations and leaf biochemicals, which has potential ramifications for animal nutrition, herbivory and litter decomposition rates. We investigated the interacting effects of atmospheric CO₂ concentrations (270, 320, 370, and 420 ppmv), plant age (42, 57, 75, and 87 days), and elevation ecotype (salt desert, sagebrush steppe, and mountain brush) on aboveground tissue nutrient levels and biochemistry of cheatgrass (*Bromus tectorum*), an important range grass in the Great Basin. Most nutrients were affected by significant ($P < 0.05$) interactions between CO₂ level and plant age, and plant ecotype and plant age. At 87 days growth, tissue C:N ratios increased significantly and concentrations of P, K, and Mg declined, with rising CO₂ levels suggesting declining forage nutrition. Tissue concentrations of Mn, K, Mg, and Ca increased with plant age and, in general, the low elevation ecotype had greater tissue nutrient concentrations than the high elevation ecotype. Hemicellulose concentration was influenced by a significant CO₂ level by ecotype interaction; overall, the high elevation ecotype had greater concentrations of hemicellulose, which increased with increasing CO₂ levels. The high elevation ecotype had significantly less acid detergent fiber than the low or mid elevation ecotypes. These data suggest that increasing atmospheric CO₂ levels may have a profound effect on the nutritional value of cheatgrass forage, and this effect may differ among elevational ecotypes.*

INTRODUCTION

Plant stoichiometry has been altered by rising atmospheric CO₂ and meta-analysis reveals that, for the important crop plants wheat and rice, tissue concentrations of many nutrients have declined (Loladze 2002). Although growth at elevated CO₂ concentration can decrease tissue nutrient concentrations, enhanced plant growth compensates such that nutrient uptake often increases (Overdieck 1993). For the important nutrient N, most research has documented decreased tissue concentrations with increased atmospheric CO₂ levels (Cotrufo and others 1996). Declining tissue concentrations are not limited to N. Indeed, comparison of nutrient concentrations from a wide variety of herbarium specimens collected during the last 250 years, indicate that recent concentrations of Al, Ca, Cu, Sr, Fe, P, Mg, Mn, K, Na, S, and Zn are invariably lower than earlier collections (Penuelas and Matamala 1993). Atmospheric CO₂

mitigated alteration of plant stoichiometry has potential ramifications for human nutrition (Loladze 2002), animal nutrition (Morgan and others 2004), herbivory (Lindroth and Dearing 2005), litter decomposition rates (Cotrufo and others 1994; Lambers 1993) and vegetation combustibility (Blank and others 2006).

In western North America, the exotic annual grass cheatgrass (*Bromus tectorum*) has aggressively invaded semi-arid to arid plant communities. The resulting increase in wildfires and shortened fire return intervals has promulgated near monocultures of cheatgrass (D'Antonio and Vitousek 1992). The large-scale invasion of the intermountain west by cheatgrass has created a sobering reality; it is now an important and seasonally nutritious forage species for native and non-native grazers (DeFlon 1986; Murray and others 1978). Like many weedy species, growth of cheatgrass is greatly stimulated by rising atmospheric CO₂ (Smith and others 1987; Ziska and others 2005). Our purpose was to determine the interacting effect of atmospheric CO₂ concentrations, plant age, and elevation ecotype on aboveground tissue nutrient levels of cheatgrass. Our working null hypotheses are:

- 1) Tissue nutrient concentrations will not vary with atmospheric CO₂ concentrations;
- 2) Tissue nutrient concentrations will not vary with plant age;
- 3) Tissue nutrient concentrations will not vary with elevational ecotype;
- 4) Nutrient uptake will not vary with atmospheric CO₂ concentrations

MATERIALS AND METHODS

Hypothesis testing began by collecting seeds from three elevation ecotypes of cheatgrass (*Bromus tectorum* L.) from northern Nevada, U.S.A. The low elevation site (low) was at 1,120 m (4,000 feet) at Poker Brown flat just west of Rye Patch Reservoir, NV (40.28° N, 118.19° W). This site receives about 16 cm (6.3 inches) of annual precipitation and is a salt desert community dominated by *Atriplex confertifolia* Torr. and Frém. S. Watson (shadscale). This site is representative of the most recent invasion by cheatgrass, which occurred in the early 1980s. In 1999, a cheatgrass-fueled wildfire consumed this site. Seeds were collected in 2000. The middle elevation site (mid) is at

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Robert R. Blank and Tye Morgan, USDA-ARS, Exotic and Invasive Weed Research Unit, Reno, NV.

²Lewis H. Ziska, USDA-ARS, Crop Systems and Global Change Lab, Beltsville, MD.

³Robert H. White, USDA-FS, Forest Products Lab, Madison, WI

1,585 m (5,200 feet) near Emigrant Pass, NV (40.38° N, 116.18° W) and receives on average 30 cm (11.8 inches) of yearly precipitation. This site has historically been dominated by *Artemisia tridentata* ssp. *vaseyana* Nutt. (mountain big sagebrush) and perennial bunchgrasses and is typical of areas where cheatgrass invasion has fostered immense landscape wildfires so detrimental to sagebrush ecosystems. This site burned in 1999 and seeds of cheatgrass were collected in 2000. The high elevation site (high) is at 2,170 m (7,120 feet) at Fox Mountain, NV (41.00° N, 119.33° W). This high elevation community receives on average about 40 cm (15.7 inches) annual precipitation and is dominated by *A. tridentata* ssp. *vaseyana*, *Pseudoroegneria spicata* (Pursh) A. Love (bluebunch wheatgrass) and *Cercocarpus ledifolium* Nutt. (curl-leaf mountain mahogany). These high elevation communities are somewhat resistant to cheatgrass invasion. Seeds were collected in 2000.

Seed lots were shipped to Dr. Lewis Ziska's CO₂ controlled environment facility at USDA-ARS, Beltsville MD. More detail on experimental protocols can be found in Ziska and others (2005). Briefly, two to three seeds were sown in pots filled with a 1:1 mixture of sand and vermiculite and thinned to one seedling 4 to 6 days after emergence. A pot volume of 22.1 L was used. For each CO₂ treatment, pots were watered to the drip point daily with a complete nutrient solution containing 14.5 mM m⁻³ nitrogen. Plants were grown at four different CO₂ concentrations (270, 320, 370 and 420 ppmv) for 24 h per day. For all chambers temperature was altered in a diurnal fashion from an overnight low of 15°C to a maximum afternoon value of 25°C, with an average daily (24 h) value of 18.3°C. Similarly photosynthetically active radiation (PAR) was also altered concurrently with temperature, with the highest PAR value (900 to 1000 :mol m² s⁻¹) occurring during the afternoon (1200 to 1500). Replicated plants were destructively harvested at 42, 57, 75, and 87 DAS (days after sowing).

Following harvest, aboveground tissue was dried to 60°C and weight recorded. Samples were shipped to the Reno, NV, USDA-ARS plant and soils laboratory for further processing. Tissue was milled separately using a 1-mm sieve opening. A portion of the milled tissue was dried for 48 hours in a desiccator and analyzed for the following. Quantification of tissue C used the Walkley-Black procedure (Walkley and Black 1947), which consists of digestion with acidified K₂Cr₂O₇ with supplemental heating to 180°C and back titration of unreacted dichromate with

FeSO₄. Tissue N was determined using micro-kjeldahl method of catalyzed oxidation in sulfuric acid and H₂O₂ (Isaac and Johnson 1976). Ammonium released was quantified using the total N module on the Lachat autoanalyzer (Salicylate method). Ashing at 550°C followed by solubilization in 1N HCl was used to measure tissue P, K, Ca, Mg, and Na (Campbell and Plank 1998). Ortho-P was quantified using a total P module (vanomolybdate chemistry) for the Lachat autoanalyzer system. Calcium, Mg, Fe, and Mn were quantified using atomic absorption spectrophotometry. Sodium and K were quantified using atomic emission spectroscopy.

Another portion of cheatgrass tissue was sent to the USDA-FS, Forest Products Lab, Madison WI for analyses of compositional carbohydrates. These analyses were only done on cheatgrass grown for 87 days. The standard method involved hydrolysis in H₂SO₄, purification, separation and quantitation using anion exchange chromatography and pulsed amperometric detection (Davis 1998).

Data structure is as follows: 4 CO₂ treatments x replication in 2 bays x 3 elevation ecotypes x 5 replications = 120 total samples. Most attributes were analyzed using a three-way ANOVA with [CO₂], ecotype, and plant age as the classification variables. Hemicellulose, acid detergent fiber, and compositional carbohydrates were only determined on cheatgrass grown for 87 days and thus were analyzed using a two-way ANOVA with [CO₂] and ecotype as classification variables. Mean comparisons used Tukey's Honest Significant Difference at the $P < 0.05$ level.

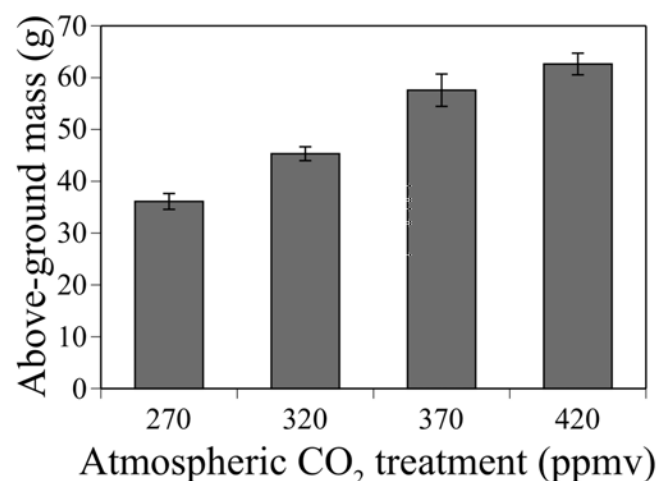


Figure 1—Relationship of above-ground cheatgrass mass with CO₂ treatment. Data are for plants grown to 87 days and pooled over ecotype. For more details, consult Ziska and others 2005.

Table 1—Result of statistical analyses for tissue nutrient concentrations in cheatgrass.

	N	C	C:N	P	K	Mg	Ca	Mn
Ecotype	0.0559	0.3421	<0.0001	<0.0001	0.0192	<0.0001	<0.0001	0.2449
CO₂	0.0094	0.0900	<0.0001	0.5932	0.3004	<0.0001	0.2036	0.6240
Ecotype x CO₂	0.8687	0.3377	0.6600	0.5313	0.9118	0.7119	0.0600	0.5307
Age	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
Ecotype x Age	0.1024	0.3416	0.0276	0.0009	0.0004	0.0089	<0.0001	0.0001
CO₂ x Age	0.0087	<0.0001	0.0175	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001
Ecotype x CO₂ x Age	0.1370	0.1854	0.3784	0.9403	0.2618	0.9878	0.1640	0.2409

RESULTS AND DISCUSSION

Growth of cheatgrass is greatly enhanced with increasing CO₂ levels (figure 1, Ziska and others 2005). Elevated CO₂ and other aspects of global change may confer a competitive advantage to some invasive plants (Dukes and Mooney 1999). Concentration of N and C in leaf tissue of cheatgrass was affected by significant CO₂ treatment by plant age interactions (table 1). Tissue N generally declined with increasing plant age, but more so for the 420 ppmv CO₂ treatment (table 2). Plants harvested on day 57 and day 87, trended lower in tissue N with exposure to greater atmospheric CO₂. Interpretation of these data is problematic in that plants were supplied luxury levels of N in order to express maximum growth. None-the-less, cheatgrass tissue N declined significantly from 4.68 percent at pre-industrial levels to 4.20 at present CO₂ levels. Our data confirms that elevated CO₂ can increase nutrient use efficiency of N (Drake and others 1997), but is affected by plant age and CO₂ level. We suspect that if this experiment were conducted in a soil medium without luxury levels of N availability, the effect may have been more pronounced. Percent C in cheatgrass tissue declined significantly in plants grown for 87 days relative to plants harvested earlier (table 2). Moreover, for plant harvest at 87 days, percent C increased significantly at the 420 ppmv CO₂ treatment compared to the other CO₂ treatments. Carbon to nitrogen ratios were affected by significant CO₂ treatment by plant age and ecotype by plant age interactions (table 1). For plants harvested at 87 days, C:N ratios increased significantly from plants grown at pre-industrial CO₂ levels to plants grown at present and future CO₂ levels (table 2). With increasing atmospheric CO₂, plants often increase tissue C:N ratios (Cotrufo and others 1996), which may reduce palatability and affect plant-herbivore interactions (Bazzaz 1990).

All other nutrients measured in cheatgrass tissue were influenced by significant CO₂ treatment by plant age and ecotype by plant age interactions (table 1). At pre-industrial and present levels of atmospheric CO₂, tissue P increased significantly for plants harvested at 87 days relative to plants harvested earlier (table 3). Tissue P declined significantly with increasing CO₂, but only

Table 2—Tissue Nutrient concentrations of C and N and C:N ratios for cheatgrass tissue.^a

CO ₂	Harvest time			
	Day 42	Day 57	Day 75	Day 87
Level/ecotype	Tissue N (%)			
270 ppmv	5.0 ^{a-c}	4.7 ^{a-b}	4.3 ^{c-e}	4.8 ^{a-d}
320 ppmv	5.2 ^{ab}	4.9 ^{a-d}	4.3 ^{c-e}	4.8 ^{a-d}
370 ppmv	4.5 ^{b-e}	4.3 ^{c-e}	4.0 ^e	4.2 ^{de}
420 ppmv	5.4 ^a	4.2 ^{de}	4.3 ^{c-e}	4.2 ^{de}
	Tissue C (%)			
270 ppmv	34.6 ^{ab}	34.7 ^{ab}	34.4 ^{ab}	31.2 ^c
320 ppmv	33.7 ^b	33.5 ^b	33.9 ^b	33.7 ^c
370 ppmv	34.9 ^{ab}	35.7 ^a	34.6 ^{ab}	33.9 ^c
420 ppmv	34.4 ^{ab}	34.0 ^b	35.2 ^{ab}	33.6 ^b
	Tissue C:N			
270 ppmv	8.2 ^{b-e}	8.6 ^{b-e}	9.2 ^{a-c}	7.6 ^{de}
320 ppmv	7.4 ^e	8.0 ^{de}	9.0 ^{a-d}	8.0 ^{ce}
370 ppmv	9.1 ^{a-d}	9.7 ^{ab}	10.1 ^a	9.2 ^{a-c}
420 ppmv	7.6 ^{de}	9.5 ^{ab}	9.0 ^{b-e}	9.2 ^{a-c}
Low	8.4 ^{bc}	9.7 ^a	9.7 ^a	9.2 ^{ab}
Mid	7.9 ^c	8.8 ^{a-c}	8.7 ^{a-c}	8.6 ^{a-c}
High	7.9 ^c	8.3 ^{bc}	9.6 ^a	7.8 ^c

^aFor each attribute, values with different superscripts are significantly different at the .05 level of probability.

for plants harvested at 87 days. At 57 days of growth, tissue P concentrations declined from the low elevation to high elevation ecotypes, but trend was not significant. With increasing plant age, concentrations of K in cheatgrass tissue increased for all CO₂ treatments and for all ecotypes (table 3). With increasing CO₂, tissue K declined significantly, but only for plants harvested on day 87. Overall, the high elevation ecotype had significantly less tissue K than the mid or low elevation ecotypes. For harvest times of 57, 75, and 87 days, tissue Mg generally declined with increasing CO₂ concentrations. For plants grown at 270 and 370 ppmv CO₂, tissue Mg generally increased with plant age. Overall, the high elevation ecotype had significantly less tissue Mg than the low or

mid-elevation ecotypes. Tissue Ca generally declined with increasing CO₂, but only significantly for plants harvested on days 57 and 75. Levels of Ca in cheatgrass tissue responded erratically with plant age, generally increasing to day 75 then declining. The high elevation ecotype had significantly more tissue Ca (47.7 mmol kg⁻¹) than either the low (57.3) or the mid elevation (53.7) ecotypes. Calcium is known to mitigate salt tolerance in plants (Marschner 1997), which may explain higher levels in the low elevation ecotype taken from a salt-desert environment. With increasing plant age, tissue Mn increased significantly (table 3). Tissue Mn varied inconsistently with CO₂ treatment and increased significantly at 420 ppmv CO₂ only for plants harvested on day 75. The high elevation ecotype had generally lower tissue Mn than the other ecotypes, but only significantly lower for plants harvested on day 87.

Hemicellulose in cheatgrass tissue was affected by a significant interaction between ecotype and CO₂ treatment (table 4). For the high elevation ecotype, hemicellulose declined by increasing CO₂ treatment. A main effect due to ecotype influenced acid detergent fiber (ADF); ADF was significantly less for the high elevation ecotype. Lack of

sample size precluded determination of interactive effects for K-lignin and composition carbohydrates. K-lignin concentration in cheatgrass tissue was significantly less for the high elevation ecotype and significantly less when grown at 320 ppmv CO₂ (table 4). Glucan was the major composition carbohydrate in cheatgrass tissue, and was significantly less in the high elevation ecotype and significantly greater in plant growth at present and future levels of CO₂.

SUMMARY AND IMPLICATIONS

Cheatgrass tissue nutrient concentrations were significantly affected by an interaction among atmospheric CO₂ concentration, plant age, and elevational ecotypes thereby refuting working null hypotheses. Our work indicates future projected increases in atmospheric CO₂ concentrations will further increase the maximum growth potential of cheatgrass, will increase its nutrient use efficiency, and alter its biochemical composition. Clearly, ecotypes of cheatgrass will respond differently to rising CO₂. Undoubtedly, the response of cheatgrass to rising CO₂ will alter its competitive profile and interactions with herbivores.

Table 3—Cheatgrass tissue nutrient concentrations of P, K, Mg, Ca, and Mn.^a

CO ₂ x Age Interaction					Ecotype x Age Interaction				
CO ₂ Level	Day 42	Day 57	Day 75	Day 87	Ecotype	Day 42	Day 57	Day 75	Day 87
Tissue P (mol kg⁻¹)									
270 ppmv	0.19 ^{bc}	0.17 ^{b-e}	0.18 ^{b-d}	0.24 ^a	Low	0.18 ^{a-c}	0.18 ^{a-c}	0.16 ^{bc}	0.19 ^{ab}
320 ppmv	0.19 ^{bc}	0.194 ^b	0.17 ^e	0.19 ^{bc}	Mid	0.18 ^{a-c}	0.17 ^{bc}	0.17 ^c	0.20 ^a
370 ppmv	0.16 ^{c-e}	0.15 ^{de}	0.14 ^e	0.19 ^{bc}	High	0.18 ^{a-c}	0.16 ^c	0.16 ^c	0.20 ^a
420 ppmv	0.17 ^{b-e}	0.16 ^{b-e}	0.16 ^{c-e}	0.17 ^{b-e}					
Tissue K (mol kg⁻¹)									
270 ppmv	1.10 ^{e-g}	1.16 ^{d-f}	1.28 ^{b-e}	1.56 ^a	Low	1.02 ^f	1.21 ^{c-e}	1.22 ^{cd}	1.50 ^a
320 ppmv	1.19 ^{d-e}	1.22 ^{c-f}	1.19 ^{d-f}	1.40 ^{ab}	Mid	1.21 ^{c-e}	1.22 ^{cd}	1.32 ^{bc}	1.44 ^{ab}
370 ppmv	0.96 ^g	1.08 ^{fg}	1.20 ^{d-f}	1.39 ^{a-c}	High	1.06 ^{ef}	1.03 ^f	1.12 ^{d-f}	1.32 ^{bc}
420 ppmv	1.13 ^{e-g}	1.15 ^{d-f}	1.21 ^{d-f}	1.31 ^{b-d}					
Tissue Mg (mol kg⁻¹)									
270 ppmv	0.17 ^{b-e}	0.20 ^{ab}	0.20 ^{ab}	0.23 ^a	Low	0.18 ^{a-d}	0.18 ^{a-c}	0.17 ^{b-d}	0.21 ^a
320 ppmv	0.19 ^{b-d}	0.20 ^{a-c}	0.16 ^{c-e}	0.18 ^{b-d}	Mid	0.17 ^{b-d}	0.19 ^{ab}	0.19 ^{a-c}	0.18 ^{a-c}
370 ppmv	0.14 ^e	0.16 ^{de}	0.17 ^{d-e}	0.20 ^c	High	0.16 ^{cd}	0.15 ^d	0.16 ^{b-d}	0.19 ^{a-c}
420 ppmv	0.17 ^{b-e}	0.15 ^{de}	0.16 ^{de}	0.16 ^{c-e}					
Tissue Ca (mmol kg⁻¹)									
270 ppmv	50.9 ^{b-d}	76.9 ^a	68.2 ^a	56.6 ^b	Low	41.6 ^e	62.1 ^{ab}	61.8 ^{ab}	64.5 ^a
320 ppmv	42.1 ^{de}	53.0 ^{bc}	53.5 ^{bc}	51.1 ^{b-d}	Mid	46.0 ^{de}	62.8 ^{ab}	58.8 ^{a-c}	47.8 ^{de}
370 ppmv	45.1 ^{c-e}	54.7 ^{bc}	56.8 ^{bc}	49.9 ^{b-d}	High	43.5 ^e	50.6 ^{c-d}	54.1 ^{b-d}	44.8 ^e
420 ppmv	36.7 ^e	49.3 ^{b-d}	54.4 ^{bc}	52.0 ^{b-d}					
Tissue Mn (mmol kg⁻¹)									
270 ppmv	1.93 ^{ef}	2.20 ^{d-f}	2.57 ^{b-e}	3.17 ^{ab}	Low	2.01 ^f	2.40 ^{c-f}	2.59 ^{c-f}	3.50 ^a
320 ppmv	2.59 ^{b-d}	3.26 ^a	2.52 ^{b-e}	3.48 ^a	Mid	2.12 ^{ef}	2.57 ^{c-f}	2.85 ^{b-d}	3.39 ^{ab}
370 ppmv	1.67 ^f	1.93 ^{ef}	2.28 ^{d-f}	3.11 ^{ab}	High	2.30 ^{ef}	2.24 ^{ef}	2.36 ^{d-f}	2.93 ^{bc}
420 ppmv	2.39 ^{c-e}	2.22 ^{d-f}	3.04 ^{ab}	3.33 ^a					

^aFor each interaction of each attribute, values with different superscripts are significantly different at the 0.05 level of probability.

Table 4—Results of statistical analyses for tissue biochemical component concentrations.^a

Table 1. Results of statistical analyses for above-ground chemical component concentrations.																	
CO ₂ ppmv	Hemicellulose			ADF		K-lignin		Glucan			Mannan						
	Ecotype			Ecotype		Ecotype		Ecotype		Ecotype		Ecotype					
	4000	5200	7120	Ecotype	Ecotype	CO ₂	Ecotype	CO ₂	Ecotype	CO ₂	Ecotype	CO ₂					
	-----%-----			%		%		%		%		%					
270	21.3 ^{b-d}	21.4 ^{bc}	24.0 ^a	Low	21.2 ^a	Low	7.7 ^{ab}	270	8.7 ^a	Low	21.9 ^a	270	20.0 ^b	Low	0.12 ^b	270	0.22 ^a
320	22.2 ^{a-c}	21.2 ^{b-d}	22.4 ^{a-c}	Mid	21.3 ^a	Mid	8.8 ^a	320	6.3 ^b	Mid	22.2 ^a	320	20.7 ^b	Mid	0.21 ^a	320	0.10 ^b
370	20.9 ^{cd}	21.9 ^{a-c}	21.3 ^{b-d}	High	18.6 ^b	High	6.6 ^b	370	8.2 ^a	High	20.1 ^b	370	22.6 ^a	High	0.13 ^b	370	0.16 ^{ab}
420	19.3 ^d	23.2 ^{ab}	21.9 ^{a-c}					420	7.7 ^a			420	22.3 ^a			420	0.13 ^b
ANOVA	Ecotype		0.0185	<0.0001		0.0243		0.0158		0.0659							
	CO2		0.4688	0.5832				0.0090		0.0069						0.0085	
	Interaction		0.0146	0.7029													

^aValues determined on cheatgrass tissue after 87 days growth. For K-lignin, glucan, and mannan, insufficient tissue for some samples only allowed comparison of main effects due to ecotype and CO₂ concentrations. ADF refers to acid detergent fiber. For each attribute, values with different superscripts are significantly different at the 0.05 level of probability.

REFERENCES

- Bazzaz, F.A. 1990. The response of natural ecosystems to the rising global CO₂ levels. *Annual Review of Ecology and Systematics*. 21:167-196.
- Blank, R.R., White, R.H., Ziska, L.H. 2006. Combustion properties of *Bromus tectorum* L.: influence of ecotype and growth under four CO₂ concentrations. *International Journal of Wildland Fire*. 15:227-236.
- Campbell, C.R., Plank, C.O. 1998. Preparation of plant tissue for laboratory analysis. *In: Kalra Y.P. (ed) Handbook of Reference Methods for Plant Analysis*. CRC Press. Pp. 37-49. Boca Raton, FL.
- Cotrufo, M.F., Ineson, P., Rowland, A.P. 1994. Decomposition of tree leaf litters grown under elevated CO₂: Effect of litter quality. *Plant and Soil*. 163:121-130.
- Cotrufo, M.F., Ineson, P., Scott, A.Y. 1996. Elevated CO₂ reduces the nitrogen concentration of plant tissues. *Global Change Biology*. 4:43-54.
- D'Antonio C.M., Vitousek, P.M. 1992. Biological invasion by exotic grasses, the grass/fire cycle, and global change. *Annual Reviews in Ecology and Systematics*. 23:63-87.
- Davis, M.W. 1998. A rapid modified method for compositional carbohydrate analysis of lignocellulosics by HPAEC/PAD. *Journal of Wood Chemistry and Technology*. 18: 235-252.
- DeFlon, J.G. 1986. The case for cheat grass. *Rangelands* 8:14-17.
- Drake, B.G., González-Meler, M.A., Long, S.P. 1997. More efficient plants: A consequence of rising atmospheric CO₂? *Annual Review of Plant Physiology and Plant Molecular Biology*. 48:609-639.
- Dukes, J.S., Mooney, H.A. 1999. Does global change increase the success of biological invaders? *Trends in Ecology and Evolution*. 14:135-139.
- Isaac, R.A., Johnson, W.C. 1976. Determination of total nitrogen in plant tissue using a block digester. *Journal of Association of Official Analytical Chemists*. 59:98-100.
- Lambers, H. 1993. Rising CO₂, secondary plant metabolism, plant-herbivore interactions and litter decomposition: Theoretical considerations. *Plant Ecology*. 104-105:263-271.
- Lindroth, R.L., Dearing, M.D. 2005. Herbivory in world of elevated CO₂. pp. 468-486. In J.R. Ehleringer, T.E. Cerling, and M.D. Dearing (eds.). *A history of atmospheric CO₂ and its effects on plants, animals, and ecosystems*. Ecological Studies 177. Springer, New York.
- Loladze, I. 2002. Rising atmospheric CO₂ and human nutrition: toward globally imbalanced plant stoichiometry? *Trends in Ecology and Evolution*. 17:457-461.
- Marschner H. 1997. Mineral nutrition of higher plants. Academic Press, London. 889 pp.
- Morgan, J.A., Mosier, A.R., Milchunas, D.G., LeCain, D.R., Nelson, J.A., Parton, W.J. 2004. CO₂ enhances productivity, alters species composition, and reduces digestibility of shortgrass steppe vegetation. *Ecological Applications*. 14:208-219.
- Murray, R.B., Mayland, H.F., Van Soest P.J. 1978. Growth and nutritional value to cattle of grasses on cheatgrass range in southern Idaho. *USDA-Forest Service Research Paper No. INT-199*. 57 pp.
- Overdieck, D. 1993. Elevated CO₂ and the mineral content of herbaceous and woody plants. *Plant Ecology*. 104-105:403-411.
- Penuelas, J., Matamala, R. 1993. Variations in the mineral composition of herbarium plant species collected during the last three centuries. *Journal of Experimental Botany*. 44:1523-1525.
- Smith, S.D., Strain, B.R., Sharkey, T.D. 1987. Effects of CO₂ enrichment on four Great Basin grasses. *Functional Ecology*. 1:139-143.
- Walkley, A., Black, I.A. 1947. Determination of organic matter in soil by chromic acid digestion. *Soil Science*. 63:251-264.
- Ziska, L.H., Reeves, J.B., Blank, B. 2005. The impact of recent increases in atmospheric CO₂ on biomass production and vegetative retention of cheatgrass (*Bromus tectorum*): implications for fire disturbance. *Global Change Biology*. 11:1325-1332.

Influence of Plant Invasion on Seed Chemistry of Winterfat, Green Rabbitbrush, Freckled Milkvetch, Indian Ricegrass and Cheatgrass

Robert R. Blank¹, Tye Morgan¹, and Fay Allen¹

ABSTRACT

Plant invasions have proven detrimental to numerous ecosystem processes; however, limited information exists on how plant invasions affect seed nutrients. We quantified nutrients in seeds of Indian ricegrass (Achnatherum hymenoides), green rabbitbrush (Chrysothamnus viscidiflorus), winterfat (Krascheninnikovia lanata), freckled milkvetch (Astragalus lentiginosus), and cheatgrass (Bromus tectorum) in sites invaded about 10 years by cheatgrass and in nearby sites with only widely scattered plants of cheatgrass. Seed chemistry differed significantly among the species tested. Overall, seeds of shrubs and freckled milkvetch had greater concentrations of N, P, and K, and lower C:N ratios than the grass species. On areas invaded by cheatgrass for 10 years, seeds trended towards decreased nutrients relative to seeds from non-invaded areas. Statistically, however, only winterfat, whose seeds from invaded areas had greater N and significantly lower C and C:N ratios, and cheatgrass, whose seeds from invaded areas had less P, were significantly different from non-invaded areas. Complimentary data suggests that cheatgrass invasion increases the availability of N, which explains an increase for seeds of winterfat; however, invasion also fosters high winterfat mortality. We suspect that high density of cheatgrass and cheatgrass litter on sites invaded for 10 years essentially ties-up large quantities of P, K and Mg, thus reducing amounts available for plant uptake. Overall, our data suggests declining nutritional value of seeds with plant invasion.

INTRODUCTION

One of the most researched invasive plant species in western rangelands has been cheatgrass (*Bromus tectorum* L.). Extensive information has been published concerning this species and how it has irreversibly changed plant communities (Mack 1981; Stewart and Hull 1949; Young and Evans 1973; Young and Blank 1995). Despite the immense pool of research that has been conducted on cheatgrass autoecology, there is an absence of literature concerning if cheatgrass invasion can alter seed nutrition of native plants. Invasive plants are known to alter Soil nutrient availability, which often facilitates their invasion (Ehrenfeld 2003). Moreover, invasive plants may alter biogeochemical cycling relative to native species such that the vertical distribution of soil nutrients is affected (Jobbágy and Jackson 2001). Modification of the vertical distribution of soil nutrient availability will affect root competition and efficacy of nutrient uptake and seed

chemistry. The purpose of this study was to quantify the effect of cheatgrass invasion of a native winterfat (*Krascheninnikovia lanata* ((Pursh) A.D.J. Meeuse and Smit) landscape on seed nutrition of the dominant species. The working null hypothesis posits that seed chemistry of several common plants in this community will not be affected by the degree of cheatgrass invasion.



Figure 1—Photographs of the study area. Top photo shows area invaded for about 10 years by cheatgrass. After 10 years of invasion cheatgrass density is incredibly high, occupies all microsites, and causes considerable mortality of winterfat (taller plant in photo). Bottom photo shows a relatively non-invaded area with healthy winterfat plants. Interspaces among winterfat are only sparsely occupied by forbs and native grasses such as squirreltail (*Elymus elymoides*), needle-and-thread (*Stipa comata*), and Indian ricegrass (center foreground in photo).

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Robert R. Blank, Tye Morgan, and Fay Allen, USDA-ARS, Exotic and Invasive Weed Research Unit, Reno, NV.

MATERIALS AND METHODS

The study site is in the Honey Lake Valley of northeastern California (40°08.18'N, 120°04.39'W). Yearly precipitation averages 230 mm (9 inches), the average maximum temperature is 18°C (64°F) and average minimum temperature is 1°C (34°F). The soil is classified as a coarse-loamy Xeric Haplocalcid. The winter-spring of 2005 to 2006, precipitation was relatively normal and provided ample moisture for the vegetation to produce an abundant source of seed. In the summer through fall of 2006, seeds were obtained when ripe from the dominant plants in a winterfat community and included cheatgrass (*Bromus tectorum* L.), winterfat (*Krascheninnikovia lanata* (Pursh) A. Meeuse and Smit), Indian ricegrass (*Achnatherum hymenoides* (Roem. and Schult.) Barkworth), green rabbitbrush (*Chrysothamnus viscidiflorus* (Hook.) Nutt.), and freckled milkvetch (*Astragalus lentiginosus* Douglas ex Hook.). Seeds were collected from 4 replicate areas (several plants per area) in two cheatgrass invasion zones: 1) invaded for 10 years by cheatgrass, and 2) only sparsely invaded in shrub canopies (figure 1). Collected seeds were allowed to air-dry in paper sacks for several months. Seeds were then hand-picked from chaff and plant material, and oven-dried at 60°C (140°F) for 48 hours. Seeds from each replicate were homogenized and a subsample was analyzed for total C and N using a LECO Tru-spec analyzer. Another subsample was ashed in a muffle furnace at 500°C (932°F) for 6 hours and treated with HCl and HNO₃ to solubilize the ash. Calcium, Mg, Na, K, Fe, and Mn were quantified by atomic absorption spectroscopy. Solubilized P was quantified using flow-injection methodology. Data were analyzed using ANOVA with categorical variables invasion class and species. Means were separate using Tukey's Honest Significant Difference at the $P \leq 0.05$ level.

RESULTS AND DISCUSSION

Overall, invasion status had minimal effects on seed chemistry (figure 2). For most species tested, the data trended towards lower nutrient concentration in areas invaded for about 10 years by cheatgrass. Seeds of winterfat were most affected by invasion status; seeds from cheatgrass invaded areas had significantly more N and significantly less C and C:N ratios than seeds from non-invaded areas. We suspect that elevated seed N in winterfat may be due to more N availability in soil beneath invaded areas facilitated by greater mineralization of organic-N (Blank 2008). Alternatively, winterfat plants in invaded areas are less dense due to die-off, which may allow greater N uptake per plant. Seeds of cheatgrass taken from areas invaded for over 10 years have significantly less P than seeds taken from relatively non-invaded areas. Explanation of lower seed P may involve a decrease of soil P availability due to far greater cheatgrass biomass on invaded areas. In addition, the invaded sites have far greater surface litter that may sequester phosphorus from newly growing plants.

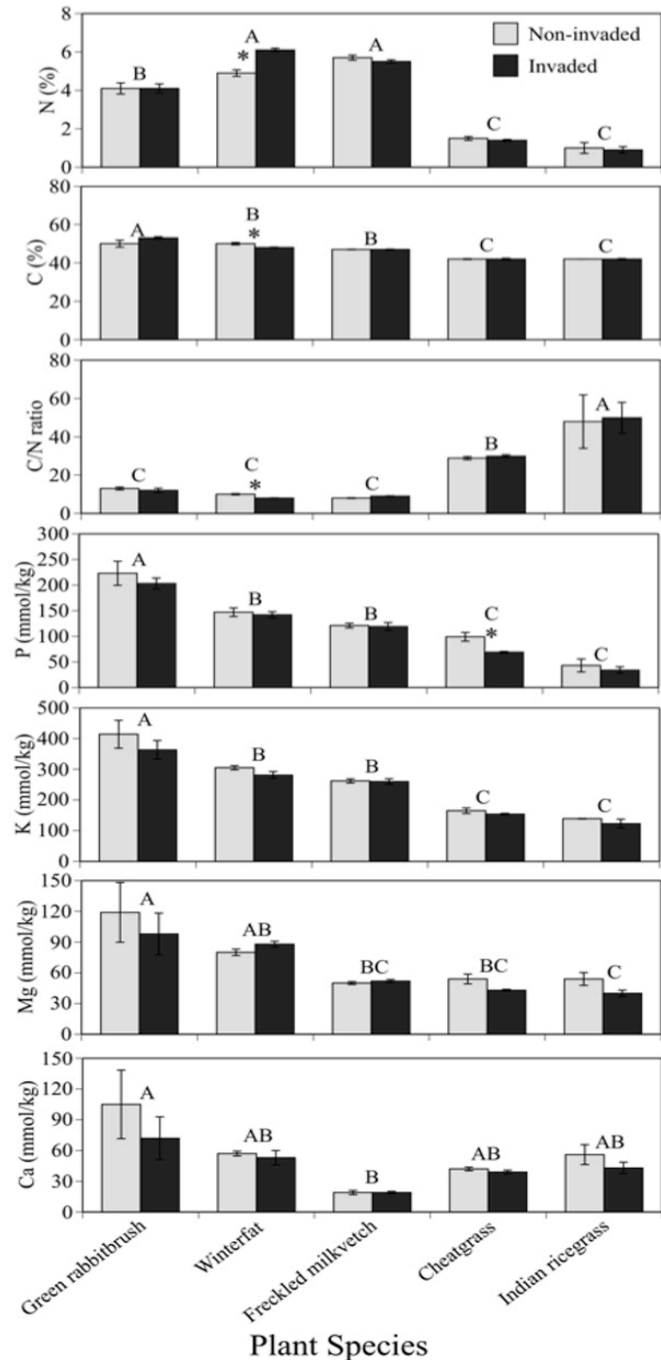


Figure 2—Nutrient concentrations in seeds of green rabbitbrush, winterfat, freckled milkvetch, cheatgrass, and Indian ricegrass as affected by invasion status. For each panel, letters denote significant differences ($P \leq 0.05$) among plants and an asterisk denotes a significant difference between non-invaded and invaded treatments.

Seeds from green rabbitbrush, winterfat, and freckled milkvetch had greater concentrations of N, P, and K and much lower C:N ratios than seeds from the grasses cheatgrass and Indian ricegrass (figure 2). At a first approximation, these data suggest seed foraging would be more efficient and profitable on a per unit weight basis.

Many other factors, however, such as caloric density and levels of secondary compounds affect foraging behavior.

SUMMARY AND IMPLICATIONS

We conclude that, in a year of slightly above normal precipitation, cheatgrass invasion had only a slight effect on nutrients in seeds of several native plants. Invasion by annual grasses such as cheatgrass and medushead (*Taeniatherum caput-medusae* L.) has altered landscape-level vegetation dynamics and has feed-back into the population structures from the soil microbial community to terrestrial fauna. Changes in biogeochemical cycling brought about by annual grass invasion has and will continue to alter the vertical distribution and availability of soil nutrient pools. Native grasses, forbs, and shrubs, which are critical forage species, will face not only competition from cheatgrass but altered nutrient niches. Ultimately, we simply lack sufficient knowledge to predict the effects of these processes on forage and seed quality for individual plant species.

REFERENCES

- Blank, R. 2008. Biogeochemistry of plant invasion: A case study with downy brome (*Bromus tectorum*). *Invasive Plant Science and Management* 1:226-238.
- Ehrenfeld, J.G. 2003. Effects of exotic plant invasions on soil nutrient cycling processes. *Ecosystems* 6:503-523.
- Jobbágy, E.G., Jackson, R.B. 2001. The distribution of soil nutrients with depth: Global patterns and imprint of plants. *Biogeochemistry* 53:51-77.
- Mack R.N. 1981. Invasion of *Bromus tectorum* L. into Western North America: An ecological chronicle. *Agro-Ecosystems* 7:145-165.
- Stewart, G., Hull, A.C. 1949. Cheatgrass (*Bromus tectorum* L.) an ecologic intruder in Southern Idaho. *Ecology* 30:58-74.
- Young, J.A., Blank, R.R. 1995. Cheatgrass and wildfires in the intermountain west. California Exotic Pest Plant Council. Symposium.
- Young, J.A., Evans, R.A. 1973. Downy brome: Intruder in the plant succession of big sagebrush communities in the Great Basin. *Journal of Range Management* 26:410-415.

Effects of Spring Prescribed Fire in Expanding Pinyon-Juniper Woodlands on Seedling Establishment of Sagebrush Species

David I. Board¹, Jeanne C. Chambers¹, and Joan G. Wright²

ABSTRACT

*Pinyon and juniper trees are expanding into mountain sagebrush communities throughout their ranges. Fire is used to restore these sagebrush communities, but limited information is available on seedling establishment of native shrubs and herbs. We examined effects of spring prescribed fire in the Great Basin on emergence and survival of five species (*Artemisia tridentata* vaseyana, *Festuca idahoensis*, *Poa secunda*, *Eriogonum umbellatum* and *Lupinus argenteus*) common to these communities. Data were collected in three microsites (undertree, undershrub and interspace) on a burned and unburned site following a prescribed fire and on the unburned site the year prior to the fire. Soil temperature and moisture were collected on both sites and years. Emergence and survival of *A. tridentata* was low. Grasses had higher emergence and survival under trees in 2003 in the unburned site, reflecting the pre-burn distribution of these species. *E. umbellatum* had high emergence and survival regardless of site or microsite. *L. argenteus* had moderate emergence that was lowest on the burned site under trees and highest on the unburned site in interspaces. Burned soils were warmer than unburned soils. Undertree microsites on the unburned site were cooler than other microsites on both sites due to shading and insulation by needle mats. Soil moisture was generally higher on the burn site due to fewer shrubs and trees. Pinyon appeared to have a facilitative role for grass seedling establishment on both sites. Spring prescribed fire did not have a negative impact on emergence or survival in these mountain sagebrush communities. Low establishment of some species indicate higher seeding rates or repeated seeding may be required. Keywords: Great Basin, sagebrush ecosystems, restoration, revegetation, seedling emergence and survival, microenvironmental conditions*

INTRODUCTION

Sagebrush communities are among the most imperiled in North America (Noss and Peters 1995; Wisdom and others 2005) and researchers and managers are actively pursuing restoration strategies to increase their sustainability. Sagebrush communities once occupied 63 million ha in western North America (West 1996; Knick and others 2003), but as much as half of this land type has been lost in the Great Basin (Noss and others 1995; Hann and others 1997) due to agricultural clearing and land development (Wisdom and others 2005), overgrazing by livestock (Knapp 1996), cheatgrass invasion and the cheatgrass-

fire cycle (D'Antonio and Vitousek 1992; Knapp 1996) and expansion of pinyon and juniper woodlands (Miller and Wigand 1994; Miller and Tausch 2002). These communities provide critical habitat for numerous sagebrush obligate species, and the loss and degradation of sagebrush communities have been linked to the decline of several species of concern including sage grouse and pygmy rabbits (Knick and others 2003).

Expansion of pinyon-juniper woodland is the major threat in moister, higher elevation areas where sagebrush communities are dominated largely by *Artemisia tridentata* ssp. *vaseyana* (Rydb.) beetle (Mountain sagebrush) (Suring and others 2005). As expanding woodlands age and canopy cover increases, the highly competitive trees can effectively eliminate the sagebrush semi-desert species in the understory. Over time the community may cross a biological threshold precluding the recovery of sagebrush communities without active restoration involving both tree removal and revegetation with sagebrush semi-desert species (Chambers 2005; Miller and others 2005). To avoid this situation managers and researchers are implementing prescribed fire projects to return landscapes to more sustainable sagebrush communities before a biological threshold is crossed (Miller and others 2005). Effective use of prescribed fire to restore and maintain sagebrush communities requires understanding fire effects on establishment of the plant species that occur in sagebrush communities, but few mechanistic studies have been conducted to date (but see Chambers and Linnerooth 2001; Wrobeliski and Kauffman 2003; Wirth and Pyke 2003).

It is often assumed that prescribed burning will promote species associated with mountain sagebrush communities, but the responses of particular species can be quite varies. Fire kills most species of sagebrush (Tisdale and Hironaks 1981), but mature tussock grasses are often resistant to burning (Wright and Klemmedson 1965). Prescribed fire has been shown to reduce competition, and increase nutrient availability (Knapp and Seastedt 1986; Dudley and Lajtha 1993). However, fire also has been shown to result in harsher environmental conditions resulting from the removal of protective trees and shrubs and higher temperatures from solar heating of blackened soil (Chambers and Linnerooth 2001). How these changes affect seedling emergence and establishment is not clear. The response is likely to differ by species and with small scale environmental variations (microsites), such as being under a tree or shrub or in the interspace.

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Board, David I., Chambers, Jeanne C., US Forest Service, Rocky Mountain Research Station, Reno, NV

²Joan G. Wright, Department of Natural Resources and Environmental Science, University of Nevada, Reno, Reno, NV

Areas under trees and shrubs often have conditions quite distinct from interspace. Soils under trees and shrubs are often moister, cooler, and higher in nutrients than the surrounding interspaces (Blackburn 1975; Doescher and others 1984; Chambers 2001). The effect of fire on the relationship between overstory and seedlings is quite complex and depends on the relative levels of competition, facilitation and the mechanisms of these processes. Fire often kills the overstory plants thus eliminating competition between the overstory and understory but also eliminating the potential for facilitation. However, some of the effects of the overstory plants remain after fire. Increased nutrients associated with the overstory plants often remain and may even increase relative to the interspace (Blank and others 1994; 2003; Rau 2005). The trees are often left standing thus providing shading albeit to a lesser degree than living trees. Shrubs are typically completely removed and thus no longer provide shade, but nutrient effects may remain. Competition for soil moisture is reduced immediately after fire and, although soil moisture deeper in the soil may increase, soil moisture near the surface may be less than pre-burn levels (Chambers and Linnerooth 2001). The lack of shading combined with soil blackening may increase evapotranspiration and increase moisture stress for young seedlings. However, the increased temperatures could increase germination rates and promote early establishment as soils heat earlier in the spring (Chambers and others 2007). This could lengthen the germination and establishment window between winter and summer.

This study examined the effects of a spring prescribed fire in a pinyon-juniper woodland on seedling establishment (emergence and survival) of one shrub (*A. tridentata* ssp. *vaseyana*), two grasses (*Festuca idahoensis* Elmer [Idaho fescue] and *Poa secunda* J. Presl. [Sandberg bluegrass]) and two forbs (*Eriogonum umbellatum* Torr. [Sulphur-flower buckwheat] and *Lupinus argenteus* Pursh. [Silvery lupine]) that occur in mountain sagebrush communities. Because of the effects of microsites in moderating environmental conditions in general, we also examined the effect of microsite (under tree, under shrub and interspace) and how it was modified after fire. Because of the importance of soil water availability in determining seedling establishment in these semi-arid ecosystems, we also examined the effects of the prescribed fire on soil temperature and soil water content on both control (not burned) and burned areas. We asked the following questions. (1) How does spring prescribed fire affect soil temperature and water content in under tree, under shrub and interspace microsites in pinyon-juniper woodlands? (2) How does spring prescribed fire affect seedling emergence and survival in under tree, undershrub and interspace microsites? We discuss the implications of the results for the restoration of sagebrush semi-desert following prescribed fire.

METHODS

Study Area

The study is located in Underdown Canyon, a Joint Fire Sciences Program demonstration area, in the Shoshone Mountain Range in Central Nevada on the Humboldt-Toiyabe National Forest (Austin Ranger District). Underdown Canyon (39° 15' 11" N, 117° 35' 83" W) is oriented east to west and contains infrequent springs and an ephemeral stream near the top of the drainage. Annual precipitation usually occurs as winter snow and spring rains averaging 23 cm at the lower elevations and 50 cm at the upper elevations. A standing rain gage located in the study area recorded 27 and 34 cm for the water years (October to October) of the two year study. The climate is characterized by cold winters and warm summers with temperatures ranging from -7.2°C in January to 29.4°C in August in Austin, NV (48 km northeast of Underdown Canyon at an elevation of 1911 m). Soils are classified as coarse loamy mixed frigid typic haploxerolls (Rau 2005).

The demonstration area is typical of an area experiencing pinyon-juniper expansion into mountain sagebrush communities in the Intermountain West (Miller and Tausch 2002). The vegetation is dominated by mountain sagebrush, *Artemisia tridentata* ssp. *vaseyana* and *Pinus monophylla* Torr. and Flem. (Single leaf pinyon) with a small component of *Juniperus osteosperma* (Torr.) Little (Utah juniper). The understory includes the grasses *Poa secunda*, *Elymus elymoides* Swezey (Squirreltail), *Hesperostipa comata* (Trin. and Rupr.) Barkworth (Needle and thread grass, *Achnatherum thurberianum* (Piper) Barkworth (Thurber's needlegrass), *Festuca idahoensis* and *Pseudoroegneria spicata* (Pursh) A. Love (Bluebunch wheatgrass), and forbs such as *Eriogonum* spp. (Buckwheats), *Crepis acuminata* Nutt. (Tartip hawksbeard), *Phlox* spp., *Lupinus argenteus* and *Penstemon* spp. (Beardtongues) *Bromus tectorum* L. (Cheatgrass), an invasive annual grass, is not prevalent on the site.

Study Design

The basic study design was a doubly blocked split plot design. The largest blocks were located on two adjacent alluvial fans. One of the fans, the higher elevation fan at 2,225m, was burned while the other fan, the lower elevation fan at 2,195m, was not burned. Nested within each of the two fans were 4 smaller blocks referred to as plots, located in areas of intermediate tree cover (~38 percent cover). Areas within these plots were designated as different microsite types: undertree, undershrub and interspace. The higher elevation fan was burned by the USDA Forest Service on 11-14 May 2002 under favorable weather conditions (air temperature < 32°C, relative humidity > 15 percent, wind speed < 9 m*s⁻¹, and gravimetric fuel moisture ≈ 40 percent). The burn was patchy because of the high fuel moisture and created a mosaic of burned and unburned vegetation across the fan.

Because this study was part of a large-scale “restoration experiment,” cost and logistics prevented burning multiple fans with similar environmental conditions. Thus, statistical inference is limited to differences between fans, rather than between burned and unburned areas. However, given that the fans were closely matched prior to burning, differences between the two fans are presumed to be due to fire. The limitations of pseudoreplication in large scale experiments have been detailed elsewhere (Hurlbert 1984; Carpenter 1990), as has the value of these types of management-scale experiments to ecology and restoration (Carpenter 1990; Miao and Carstenn 2006)

Soil Temperature and Water Measurements

Soil temperature and gravimetric soil water on both the burned and unburned blocks were measured within three of the plots for undershrub, undertree and interspace microsites (2 blocks x 3 plots x 3 microsites = 18 locations). Soil water samples were collected every 2 to 4 weeks during the growing season in 2001 and 2002 from a soil core divided into three depths (0 to 3, 3 to 8 and 8 to 23 cm) that reflected the natural soil horizons that exist on both fans (Rau 2005). Gravimetric soil moisture was determined based on wet versus oven-dry weights. Soil temperature data were collected with networks of three interconnected thermocouples of equal length, spaced 50 cm apart and installed at depths of 2, 5 and 20 cm. Hourly averages were taken from measurements taken every 5 minutes for June through October of 2002 and April through August of 2003 using Campbell Scientific CR-10 and CR-10x microloggers and then averaged over each month.

Seedling Emergence and Survival Measurements

Commercial seed lots of five common sagebrush semi-desert were used including one shrub (*A. tridentata* ssp. *vaseyana*), two grasses (*Festuca idahoensis* and *Poa secunda*) and two forbs (*Eriogonum umbellatum* and *Lupinus argenteus*). A standard tetrazolium test (Moore 1972; Peters 2000) determined viability of seeds less than one month before seeding. Seeding grids (50 x 50 cm quadrats) were placed within each of the four plots in undertree, undershrub and interspace microsites (4 plots x 3 microsites x 5 species = 60 quadrats). Planting occurred on the unburned site in 2001 and on both the unburned site and burned site in 2002 ($n = 180$ quadrats). Undertree quadrats were placed 50 cm from the tree canopy edge towards the bole. The undershrub quadrats were located to maximize shrub canopy coverage. Interspace quadrats were placed in the forb grass matrix surrounding the trees and shrubs. Each seeding quadrat was prepared for planting by removing understory plants and surface litter and reserving the litter. One hundred filled seeds of each species were then seeded in each quadrat by planting one seed into each of 100 grid cells (5.0 x 5.0 cm). Seed of *P. secunda* and *A. tridentata* were planted at a depth of 0.5 cm because of their small

size; all others were planted at 1.0 cm. Following seeding, a one centimeter layer of the reserved litter was spread over the quadrat. Seedling emergence and survival were monitored every two weeks from April through June after planting and monthly from July through October in 2002 and 2003 on the unburned block and in 2003 on the burned block.

Statistical Analysis

Environmental data

Analysis was based on a multilocation trial experiment with locations (fans) viewed as fixed effects. Plots were treated as random samples within each fan. A separate analysis was done for each year for the soil moisture and soil temperature data. For both the soil temperature and soil moisture data, depth was treated as a split plot factor within microsite. For soil temperature, correlation between the three depths for each core was modeled as a random effect with a toeplitz covariance structure. The month of sampling for each temperature sensor was treated as a repeated measure with a toeplitz covariance structure. For soil moisture, correlation between depths for a core was assumed to be zero because of model convergence problems. Date of sampling for each core depth was modeled as a repeated measure with the covariance structure being a power function ($\sigma^2 \rho_{ij}^d$) of the days between sample dates. The best fit covariance structures were selected using AIC from those structures available in SAS Proc Mixed (SAS 2000).

Seedling Emergence and Establishment

The proportion of viable seeds that emerged from each quadrat and the proportion of those seedlings that survived until the final sampling period of the planting year (establishment) were arcsine square root transformed. The transformed emergence and establishment data also were analyzed based on a fixed multilocation trial study design. Each fan-year was treated as an independent location and requires the assumption of independence between fan-years. This allowed planned comparisons between 2002 and 2003 on the control and between the burned and unburned fan in 2003 as well as the related interaction with microsite by creating estimable functions for the particular hypothesis to be tested. Species were analyzed by functional group, (grass, forb or shrub) with species treated as a split factor within functional group. Each species was analyzed separately as well.

All statistical models were fit using SAS Proc mixed (SAS 2000). Standard checks of model assumptions were conducted and all were acceptable given the robustness of the model fitting procedures.

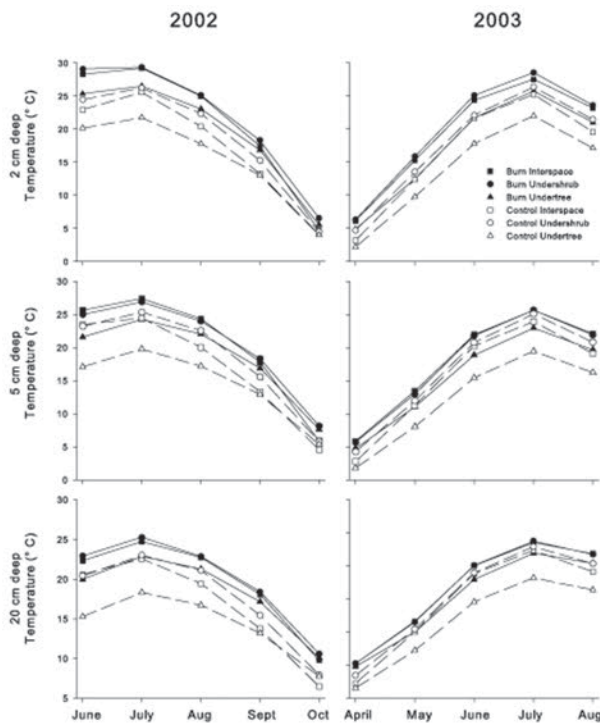


Figure 1—Average monthly temperature (°C) in the unburned and burned site for interspace, undershrub and undertree microsites at three depths, 2cm, 5cm and 20cm.

RESULTS

Effect of Site (Burned and Unburned Fan) and Microsite on Soil Moisture and Temperature

Soil temperature was highest in June, July and August with July being the warmest month in 2002 and 2003 (figure 1). The burned site soil temperatures were warmer than the unburned site in both years (table 1). The undertree microsite was cooler than the undershrub and interspace microsites on both sites and in both years. The differences between the microsites were most distinct during the warmest months sampled. The coolest locations were under trees on the unburned site in both years and all months. The undershrub and interspace microsites on the unburned site and the undertree microsites on the burned site were significantly cooler than the interspace and undershrub on the burned site in the warmest months. Soil temperature decreased with depth on both the unburned and burn sites for all microsites, but these differences were most apparent during the warmest months.

Soil moisture was highest in early spring dropping off quickly by mid-May. This pattern occurred at all depths, although the rate of decline decreased with depth and deeper soils remained wetter longer. In 2002, during the late spring the undertree microsite remained significantly wetter than other microsites on the unburned site (table 2;

figure 2). On the burned site the undertree microsites did not differ from the other microsites during this period except immediately after rain. In 2003, there were no significant differences between microsites on either site.

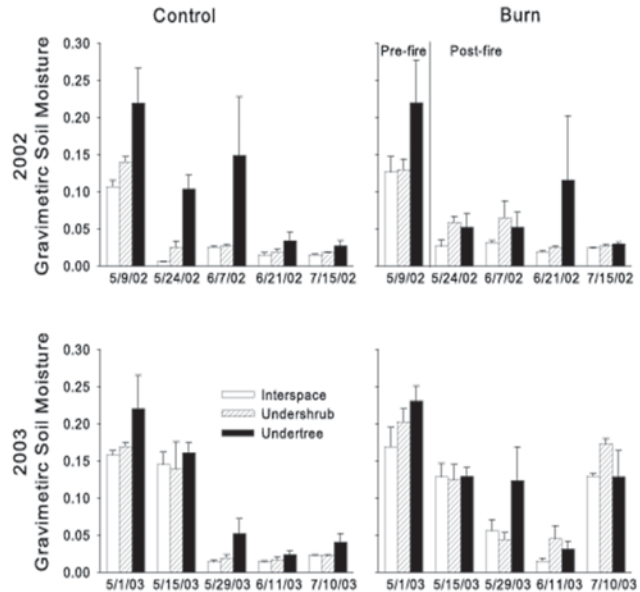


Figure 2—Average gravimetric soil moisture at the 0-3 cm depth from three samples taken on the control and burn sites in the interspace, undershrub or undertree microsites in May to Mid-July in 2002 and 2003. The burned site was burned on May 11 through the 14th 2002.

Effect of Site (Burned and Unburned Fan) and Microsite on Emergence and Survival

Emergence of the shrub *A. tridentata* was low, especially in 2003 (figure 3). The only significant difference was on the unburned site in 2002 versus 2003. Relatively high emergence occurred in the interspace in 2002 compared to the undershrub and undertree locations in 2003 (table 3; figure 3).

In 2002, none of the *A. tridentata* seedlings that emerged survived (figure 3). In 2003 *A. tridentata* seedlings that emerged on the unburned site under shrubs and on the burned site in the undertree locations did not survive. There were no significant differences among the remaining treatments.

The two grasses, *P. secunda* and *F. idahoensis*, had similar emergence patterns for year, site and microsite. In fact, if analyzed together there were no significant interactions with species, although *F. idahoensis* had significantly higher emergence ($F_{1,9} = 6.38$, $p = 0.032$; figure 3). Emergence did not differ among microsites for either *F. idahoensis* or *P. secunda* in 2002 on the unburned site. However, in 2003 the undertree microsite had higher emergence than the other two microsites for both species.

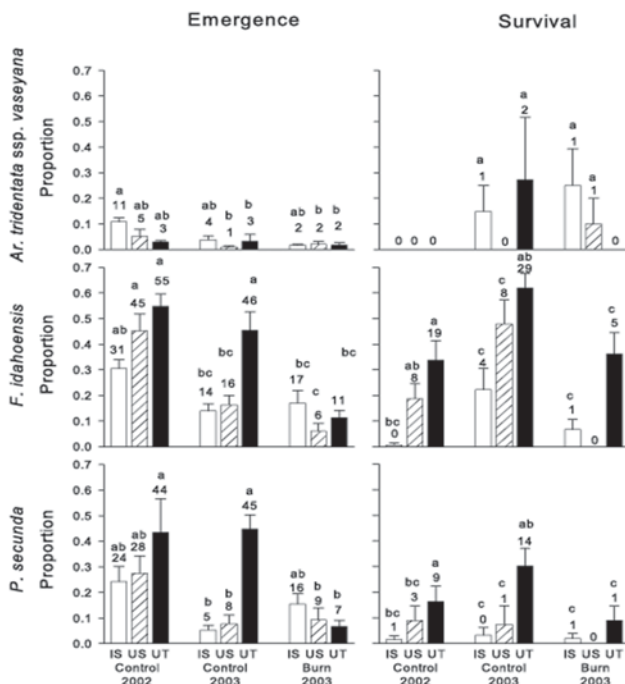


Figure 3—Proportional emergence and survival of *Ar. tridentata* ssp. *vaseyana*, *F. idahoensis* and *P. secunda* in the control site in 2002 and 2003 and burn site in 2003 for interspace (IS), Undershrub (US) and Undertree (UT) microsites. Numbers above each bar indicate the average number of individuals emerging or surviving per quadrat out of 100 seeds planted. Bars that share the same letters in each graph are not significantly different than each other at an alpha of 0.05.

For *P. secunda* and *F. idahoensis* emergence within microsites did not differ between years although there was generally lower emergence in 2003. For *P. secunda* the difference between years was primarily due to decreased emergence in the interspace and undershrub locations. The difference between the years on the unburned fan for *F. idahoensis* was much greater for the undershrub locations and if taken individually this difference was significant ($t_{1,26} = 4.04$, $p_{\text{tukey}} = 0.017$; figure 3; table 3).

The two grass species also had similar seedling survival patterns, with no significant interactions between species and microsite, year, or site. *F. idahoensis* had significantly greater survival overall than *P. secunda* ($f_{1,45} = 7.28$, $p = 0.010$; figure 3). Both species had the highest survival rates under trees on the unburned site in both years, although for *F. idahoensis* survival under trees was not significantly greater than in the undershrub areas in 2002 (table 3). Survival in 2003 was higher than in 2002 for both species on the unburned site. For *F. idahoensis* survival in undershrub locations differed between years although survival in the other microsites did not. Survival did not differ among microsites on the burned site for either species, although survival in the undertree locations remained higher than other microsites. Neither species survived on the burned site in undershrub locations (figure 3).

Table 1—Repeated measures mixed effects ANOVA for average monthly soil temperatures (month and depth were treated as repeated measure with toeplitz covariance structure). *P* values ≤ 0.05 and > 0.01 are italicized, *p* values ≤ 0.01 and > 0.001 are bold faced and *p* values ≤ 0.001 are bold faced and italicized.

Effect	2002			2003		
	<i>df</i>	F-Value	<i>P</i>	<i>df</i>	F-Value	<i>P</i>
Site	1,4	65.31	0.001	1,4	31.49	0.005
Microsite	2,8	16.26	0.002	2,8	44.66	<0.001
Microsite* Site	2,8	1.22	0.346	2,8	3.85	0.067
Depth	2,24	11.75	<0.001	2,24	60.75	<0.001
Depth*Site	2,24	1.37	0.273	2,24	1.90	0.172
Depth* Microsite	4,24	0.61	0.659	4,24	1.32	0.292
Depth* Site* Microsite	4,24	0.41	0.802	4,24	0.66	0.623
Month	4,144	2403	<0.001	4,144	2081	<0.001
Month* Site	4,144	19.25	<0.001	4,144	5.95	<0.001
Month* Microsite	8,144	10.93	<0.001	8,144	8.82	<0.001
Month* Microsite* Site	8,144	4.37	<0.001	8,144	1.29	0.254
Month* Depth	8,144	37.55	<0.001	8,144	17.55	<0.001
Month* Depth* Site	8,144	2.69	0.009	8,144	0.95	0.475
Month* Depth* Microsite	16,144	0.98	0.482	16,144	0.59	0.885
Month* Depth* Site* Microsite	16,144	0.54	0.924	16,144	0.12	1.000

Table 2—Repeated measures mixed effects ANOVA for gravimetric soil moisture (date was treated as repeated measure with a power covariance structure). *P* values ≤ 0.05 and > 0.01 are italicized, *p* values ≤ 0.01 and > 0.001 are bold faced and *p* values ≤ 0.001 are bold faced and italicized.

Effect	2002			2003		
	<i>df</i>	F-Value	<i>P</i>	<i>df</i>	F-Value	<i>P</i>
Site (Burn vs. Unburned)	1,4	15.04	<i>0.018</i>	1,4	20.38	<i>0.010</i>
Microsite	2,8	7.92	<i>0.013</i>	2,8	7.08	<i>0.017</i>
Microsite* Site	2,8	0.40	0.686	2,8	0.93	0.433
Depth	2,24	5.97	0.008	2,275	5.74	0.004
Depth*Site	2,24	2.68	0.089	2,275	1.30	0.275
Depth* Microsite	4,24	2.00	0.127	4,275	0.18	0.947
Depth* Site* Microsite	4,24	0.55	0.701	4,275	0.49	0.742
Date	9,324	262.60	<0.001	7,275	114.23	<0.001
Date* Site	9,324	4.45	<0.001	7,275	6.66	<0.001
Date* Microsite	18,324	3.41	<0.001	14,275	1.69	0.057
Date* Microsite* Site	18,324	1.75	<i>0.030</i>	14,275	2.41	0.003
Date* Depth	18,324	7.40	<0.001	14,275	4.30	<0.001
Date* Depth* Site	18,324	1.41	0.123	14,275	0.91	0.548
Date* Depth* Microsite	36,324	1.21	0.197	28,275	0.68	0.886
Date* Depth* Site* Microsite	36,324	0.73	0.870	28,275	0.30	0.999

The two forbs had quite different emergence patterns compared to the grasses. If analyzed together they also did not show any significant interactions for species although there was a significant species effect: *E. umbellatum* had higher emergence than *L. argenteus*. *E. umbellatum* analyzed alone had no significant differences in emergence between years, sites or microsites (figure 4, table 3). *L. argenteus* had no significant difference among microsites on the unburned site in 2002, but in 2003 the interspace areas had significantly higher emergence than the undershrub areas. There was no difference in emergence among different microsites on the burned site. The undertree areas on the burned site had significantly less emergence than interspace areas on the unburned site.

The two forbs were similar in their survival patterns, showing no significant interactions between species or other factors when analyzed together. *E. umbellatum* did have significantly greater survival than *L. argenteus* ($f_{1,48} = 26.21$, $p < 0.001$). *E. umbellatum* had no significant differences in survival with microsite, year, site or their interactions (table 3). *L. argenteus* did not survive in the undershrub locations on the unburned site in 2003 or the undertree locations on the burned site (figure 4).

Overwinter and Second Year Survival on Unburned Site

Seedlings that emerged in 2002 on the unburned site were followed over winter and during the 2003 growing season. Overwinter survival was high for all species ranging from 33 to 100 percent and averaging 82 percent. Overwinter survival was not significantly different between microsites for any species. The second season survival was much more

variable ranging from 0 to 85 percent with a mean of 35 percent. The only significant difference was for the grasses where undertree and undershrub microsites had higher survival than interspaces ($f_{2,8.3} = 7.11$; $p = 0.01$). Demographic stochasticity probably played a large role in the survival estimates during this period, especially in those species that had low initial emergence and survival.

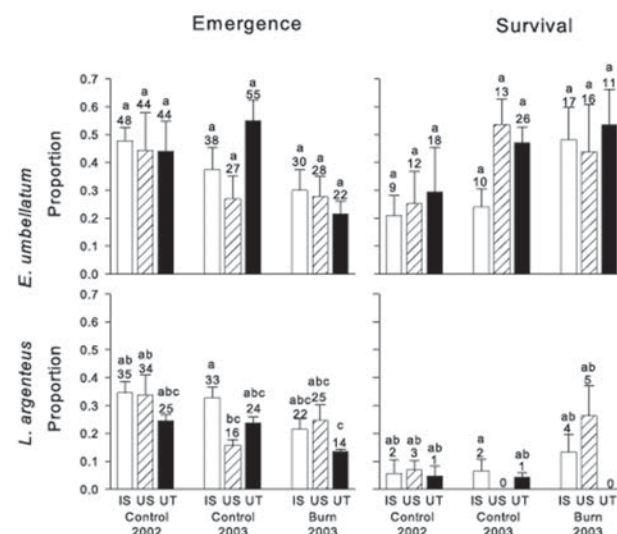


Figure 4—Proportional emergence and survival of *E. umbellatum* and *L. argenteus* in the control site in 2002 and 2003 and burn site in 2003 for interspace (IS), Undershrub (US) and Undertree (UT) microsites. Numbers above each bar indicate the average number of individuals emerging or surviving per quadrat out of 100 seeds planted. Bars that share the same letters in each graph are not significantly different than each other at an alpha of 0.05.

Table 3—ANOVA tables for arcsine square root transformed emergence and survival through the first growing season for *A. tridentata*, *P. secunda*, *F. idahoensis*, *E. umbellatum* and *L. argenteus*. Year effects were tested on the control sites and site effects (burn versus control site) were only tested in 2003. *P* value ≤ 0.05 and > 0.01 are italicized, *p* values ≤ 0.01 and > 0.001 are bold faced and *p* values ≤ 0.001 are bold faced and italicized.

Effect	<i>A.tridentata</i>			<i>P.secunda</i>		<i>F.idahoensis</i>		<i>E.umbellatum</i>		<i>L.argenteus</i>	
Emergence	df	F	<i>p</i> -value	F	<i>p</i> -value	F	<i>p</i> -value	F	<i>p</i> -value	F	<i>p</i> -value
Microsite	2,9	7.16	0.058	6.16	0.009	10.66	0.001	0.95	0.406	5.49	<i>0.014</i>
Year (Control)	1,9	9.94	0.004	5.62	<i>0.042</i>	16.84	0.003	0.46	0.515	2.81	0.128
Site (2003)	1,9	0.06	0.814	1.91	0.200	13.48	0.005	2.16	0.175	1.20	0.302
Year*Microsite	2,18	0.73	0.493	2.78	0.088	2.30	0.129	2.15	0.146	4.27	<i>0.030</i>
Site*Microsite (2003)	2,18	0.41	0.666	9.36	0.002	7.88	0.004	3.23	0.063	6.54	0.007
Survival											
Microsite	2,9	0.52	0.601	7.61	0.005	10.29	0.001	0.33	0.725	6.12	0.009

DISCUSSION

The patterns of microsite environmental conditions and seedling establishment observed in this study are consistent with most studies of fire and microsite in the Great Basin. Higher soil temperatures on the burn site especially at the surface may be attributed to several factors including increased solar radiation reaching the soil and increased soil heating from soil blackening. Higher soil temperatures have been reported after harvesting in pinyon-juniper woodlands (Everett 1985) and in burned riparian corridors dominated by *Artemisia tridentata* var. *tridentata* (Chambers and Linnerooth 2001). The relatively lower temperatures in the undertree microsites compared to the interspace and undershrub microsites on the burned site was likely caused by partial shading from standing dead trees as well as the retention of needle mats after the fire that insulated the soil. The low temperatures on the undertree microsites on the unburned site were probably due not only to the needle mats but also the increased shading provided by live trees.

The generally higher soil moisture in the burn site might be due to lower transpiration rate due to vegetation loss. Soil moisture during the growing season in burned riparian corridors dominated by big sagebrush was also higher on burned sites (Chambers and Linnerooth 2001). Soil moisture in pinyon-juniper woodlands has been reported to increase during the first 2 to 3 years after tree harvesting as well indicating that tree death or removal is driving the difference (Everett 1985). The higher soil moisture under trees during spring and early summer is probably due to lower evaporation due shading, the insulating effect of the litter mat and higher organic content of these soils compared to the undershrub and interspace locations.

Seeding success of *A. tridentata* is typically low (Young and Evans 1986; Meyer 1990; Booth and others 1999) and natural recruitment is believed to be episodic (Lommasson 1948; Cawker 1980; Perryman and others 2000). Many of the reported reasons for poor seeding success might have affected this study as well. The planting depth of 0.5 cm

was similar to that recommended for this species (broadcast seeding to < 0.5 cm; McArthur and Welch 1982; McArthur and others 1995), but in undertree and shrub microsites an additional 1 cm of litter was added that may have decreased emergence rates. Although the seeds provided by the commercial growers were from areas with similar environmental characteristics, the lack of locally-adapted populations may have resulted in mis-timed germination and caused increased mortality (Meyer and Monson 1990). We may have under estimated emergence because *A. tridentata* is expected to emerge in late winter to early spring often under snow (Meyer et al 1990; Meyer and Monson 1990) or coincident with snow melt (Monson 1999) before sampling began. Mortality can be high in the first few weeks of emergence (a period when the study site was inaccessible). This mortality has been associated with damping off (Meyer and Monson 1990) and early moisture stress (Boltz 1994). The low seedling survival of those that were detected as emerging was likely due to the combined effects of late spring frosts as observed by Meyer and Monson (1990) and low soil moisture as observed by Owens and Norton (1992). The two years sampled were dry to normal based on annual precipitation years (14.9 and 21.0 cm total for the water year 2002 and 2003 respectively compared to a 30 year mean of 20.1 cm from the Austin, NV weather station).

The emergence and survival patterns of the two grasses parallels their distribution on the landscape. Both grasses are more common under trees, especially at the edge of the litter layer (Everret and others 1983), and generally higher grass cover under trees has been reported in North American arid landscapes (Armentrout and Pieper 1988). Greater survival of grasses under shrubs has been reported for arid shrub communities (Garcia-Moya and McKell 1970; Huber-Sannwald and Pyke 2005) and, although this pattern of higher survival was seen in our data, the increase was not significant. The increase in emergence and survival under trees was not observed on the burned site. Also, neither *F. idahoensis* nor *P. secunda* survived in undershrub microsites on the burned site. Increased exposure and more

extreme soil temperatures can cause higher seedling mortality in undershrub and undertree microsites following burning (Chambers and Linnerooth 2001). However, resource availability, both nutrients (Blank and others 1994, 2003) and water (Chambers and Linnerooth 2001), typically increases following fire and can result in increased growth and reproduction of surviving seedlings (Chambers and others 2007). Growing season conditions appeared to modify both emergence and survival patterns. On the unburned fan, 2002 had better emergence than 2003. Near-surface soil temperatures averaged 10°C warmer in 2002 than in 2003 and this may have led to increased germination rates (for example, Chambers and others 2007). In contrast, survival on the unburned sites was higher in 2003, a wetter year compared to 2002.

The two forbs did not show any clear patterns in response to microsite or location (fan). *E. umbellatum* was not affected by microsite or burning and emergence and survival was high in all locations. *E. umbellatum* is common throughout both the burned and unburned site and natural seedlings were observed establishing in equal abundance on both sites (Dhaemers 2006). *L. argenteus* had fair to poor emergence and very poor survival. Emergence was higher in interspaces in 2003 than in undershrub microsites on the control site although this was not the case in 2002. The universally poor survival of *L. argenteus* observed here contrasted with an observed increase in cover and natural seedling establishment on the burned site three years later (E. Goergin University Nevada, Reno, NV, personal communication). This indicates that both locally adapted seed sources and site modification may be important for establishing *L. argenteus* following prescribed fire.

This study was conducted in plots with intermediate tree cover (38 percent) and the immediate effects of the spring prescribed burn appeared to be more negative for seedlings than adult plants. In a concurrent study on the same site there was an increase in cover of both the perennial grass and forb species on the burned site, including the species observed in this study (Dhaemers 2006). Adult plants that survived the fire were undoubtedly more tolerant of the harsh post-fire environment than seedlings, and the increase in cover came largely from pre-existing individuals. Survivors were released from competition from the shrubs and trees, and exhibited increased productivity because of an increase of available soil water and nutrients (Rau 2005). The effects of the prescribed fire on seedling emergence and survival appeared to moderate over time, and seedling establishment from reproducing adults occurred on the site within three years after the burn (Dhaemers 2006).

As expanding woodlands mature and canopy cover increase, perennial grass and forb cover typically decreases and the risk of high severity crown fire increases (Miller and Tausch 2002). On the study site areas dominated by

trees (78 percent canopy cover) had low cover and biomass of perennial herbaceous species and limited shrub establishment three years after the burn (Dhaemers 2006). Low abundance of perennial herbaceous species on these sites resulted in limited seed input (Allen and others submitted), and environmental conditions after prescribed fire were at least as harsh as those on areas with intermediate tree cover. This indicates that as tree cover increases the probability of unassisted recovery of these sites decreases and the importance of revegetation and seedling establishment increases. Sites with high tree cover may have crossed biological thresholds, and restoration to sagebrush semi-desert species following either prescribed fire or wildfire will likely require revegetation with native semi-desert species.

CONCLUSIONS

This study contributes to our basic understanding of the effects of fire on seedling establishment in semi-arid woodlands and shrublands and has important implications for their restoration. The spring prescribed fire in the study resulted in increases in soil resources, including both soil nutrients (Rau 2005) and soil water as has been observed after wildfires or fall prescribed burns in similar ecosystems (Blank and others 1994, 2003; Chambers and Linnerooth 2001). Increased exposure and blackened soils following the burn resulted in higher soil temperatures as observed elsewhere (Chambers and Linnerooth 2001), but environmental conditions differed among microsites. Shrubs and their litter usually were consumed completely by the fire resulting in a harsher microenvironment. In contrast, while trees were killed by the relatively cool spring burn, the litter mats remained intact and the standing trees coupled with the litter mats moderated the soil environment. Individual species responses to the burn differed, but seedling establishment was generally low. Grasses are often associated with trees in semi-arid ecosystems (Everett and others 1983; Armentrout and Pieper 1988), and emergence and survival of both *F. idahoensis* and *P. secunda* were lower on the burned than unburned site. In contrast, there were no detectable differences in emergence and survival on burned vs. unburned sites for *A. tridentata*, *E. umbellatum*, or *L. argenteus*. Differences in growing season conditions affected seedling emergence and survival on the unburned site and likely influenced seedling establishment on the burned site as well. An early and warm growing season in 2002 resulted in higher seedling emergence, but cooler and wetter conditions in 2003 resulted in greater seedling survival.

Several recommendations for the revegetation of sagebrush ecosystems that have crossed biological thresholds due to pinyon-juniper encroachment follow from the results of this research. Our data confirm the need for increasing the seed

availability of adapted populations of native species. Seeds of some species, like *E. umbellatum*, exhibited relatively high establishment across sites and years and appear to be fairly broadly adapted. Other species, like *L. argenteus* and *A. tridentata* ssp. *vaseyana*, exhibited generally low emergence or survival from purchased seeds and may require locally adapted ecotypes for successful establishment (see Meyer and others 1990). As observed elsewhere, seedling establishment depended on environmental conditions in these ecosystems and was highly variable among years (Chambers 2000; 2001). Seeding species at higher rates than is typical may be necessary to ensure establishment during more favorable years. Also, repeated seeding of *A. tridentata* species (which have short-lived seeds) in high priority areas may be required to promote establishment. Clearly, effective approaches for restoring mountain sagebrush ecosystems are needed to ensure their sustainability. A major focus should be on increasing the seed availability of locally-adapted species and developing methods for their establishment.

ACKNOWLEDGEMENTS

This research was funded by the US Forest Service, Rocky Mountain Research Station, Great Basin Ecosystem Management Project, and Joint Fire Sciences Program Project # O1B-3-3-01. We thank the Humboldt-Toiyabe National Forest for facilitating this research, Dave Turner for assistance with the statistical design, and Monica Mazzola, Eugenie Mont Blanc, Ben Rau and Rich Redd for field assistance.

REFERENCES

- Allen, E.A., Chambers, J.C., Nowak, R.S. 2008. Effects of a spring prescribed burn on the soil seed bank in sagebrush steppe exhibiting pinyon-juniper expansion. *Western North American Naturalist*. 68(3):265-277.
- Armentrout, S.M., Pieper, R.D. 1988. Plant distribution surrounding rocky mountain pinyon pine and one seeded juniper in south-central New Mexico. *Journal of Range Management* 41(2):139-143.
- Blackburn, W.H., 1975. Factors influencing infiltration and sediment production of semiarid rangelands in Nevada. *Water Resources Research* 11(6):929-937.
- Blank, R.R., Allen, F., Young, J.A. 1994. Extractable anions in soils following wildfire in a sagebrush-grass community. *Soil Science Society of America Journal* 58:564-570.
- Blank, R.R., Chambers, J.C., Zamudio D.C. 2003. Prescribed burning of central Nevada degraded riparian ecosystems: Effects on soil and vegetation. *Journal of Range Management* 56:387-395.
- Boltz, M. 1994. Factors influencing postfire sagebrush regeneration in south-central Idaho. Pages 281-290 in *Ecology, Management, and Restoration of Intermountain Annual Rangelands: Proceedings*, 18–22 May 1992, Boise, ID. INT-GTR-313. USDA, Forest Service, Intermountain Research Station.
- Booth, D.T., Gores, J.K., Schuman, G.E., Olson, R.A. 1999. Shrub densities on pre-1985 reclaimed mine lands in Wyoming. *Restoration Ecology* 7(1):24–32.
- Carpenter, S.R. 1990. Large-Scale Perturbations: Opportunities for Innovation. *Ecology* 71(6): 2038–2043
- Cawker, K.B. 1980. Evidence of climatic control from population age structure of *Artemisia tridentata* Nutt. in southern British Columbia. *Journal of Biogeography* 7(3):237-248.
- Chambers, J.C. 2000. Seed movements and seedling fates in disturbed sagebrush steppe ecosystems: implications for restoration. *Ecological Applications* 10(5):1400-1413.
- Chambers, J.C. 2001. Pinus monophylla establishment in an expanding pinyon-juniper woodland: Environmental conditions, facilitation and interacting factors. *Journal of Vegetation Science* 12(1):27-40.
- Chambers, J.C. 2005. Fire related restoration issues in woodland and rangeland ecosystems. Pages 149-160 in L. Taylor, et al., compilers. *Mixed Fire Regimes: Ecology and Management*. Symposium proceedings. November 17-19, 2004, Spokane, WA.
- Chambers, J.C., Linnerooth, A.R. 2001. Restoring Riparian Meadows Currently Dominated by *Artemisia* using Alternative State Concepts: The Establishment Component. *Applied Vegetation Science* 4(2):157-166.
- Chambers, J.C., Roundy, B.A., Blank, R.R., Meyer, S.E., Whittaker, A. 2007. What makes Great Basin sagebrush ecosystems invisable by *Bromus tectorum*? *Ecological Monographs* 77(1): 117-145.
- D'Antonio, C.M., Vitousek, P.M. 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global change. *Annual Review of Ecology and Systematics* 23:63–87.
- Dhaemers, J. 2006. Vegetation recovery following spring prescribed fire in pinyon-juniper woodlands of central Nevada: Effects of elevation and tree cover. MSc. Thesis. University Nevada, Reno.
- Doescher, P., Miller, S., Winward, A.H. 1984. Soil chemical patterns under eastern Oregon plant communities dominated by big sagebrush. *Soil Science Society of America Journal* 48(3):659-663.
- Dudley, J.L., Lajtha, K. 1993. The effects of prescribed burning on nutrient availability and primary production in sandplain grasslands. *American Midland Naturalist* 130(2):286–298.
- Everett, R.L. 1985. Great Basin pinyon and juniper communities and their response to management. Pages 53–61 in 8th Annual Meeting of the Society for Range Management, 11-15 February 1985. Salt Lake City, UT. The Society for Range Management. Denver, CO.
- Everett, R.L., Sharrow, S.H., Meeuwig, R.O. 1983. Pinyon-Juniper Woodland Understory Distribution Patterns and Species Associations. *Bulletin Torrey Botanical Club* 110(4):454-463
- Garcia-Moya, E., McKell, C.M. 1970. Contribution of shrubs to the nitrogen economy a desert-wash plant community. *Ecology*. 51(1):81-88.
- Hann, W.J., Jones, J.L., Karl, M.G., Hessburg, P.F., Keane, R.E., Long, D.G., Menakis, McNicoll, C.H., Leonard, Gravenmier, R.A., Smith, B.G. 1997. Landscape dynamics of the basin. Pages 337–1055 in T.M. Quigley and S.J. Arbelbide, technical editors. *An assessment of ecosystem components in the interior Columbia Basin and portions of the Klamath and Great Basins*. General technical report PNW-GTR-405. U.S. Forest Service, Pacific Northwest Research Station.
- Huber-Sannwald, E., Pyke, D.A. 2005. Establishing native grasses in a big sagebrush dominated site: an intermediate restoration step. *Restoration Ecology* 13(2):292–301.
- Hurlbert, S.H. 1984. Pseudoreplication and the design of ecological field experiments. *Ecological Monographs*. 54(2):187-211
- Knapp, P.A. 1996. Cheatgrass (*Bromus tectorum*) dominance in the Great Basin Desert. *Global Environmental Change* 6(1):37-52.
- Knapp A.K., Seastedt, T.R. 1986. Detritus accumulation limits productivity of tallgrass prairie. *BioScience* 36(10):662–8.
- Knick, S.T., Dobkin, D.S., Rotenberry, J.T., Schroeder, M., Van der Haegen, W.M., Van Riper Jr, C. 2003. Teetering on the edge of too late? Conservation and research issues for avifauna of sagebrush habitats. *Condor*. 105(4):611–634.
- Lommasson, T. 1948. Succession in sagebrush. *Journal of Range Management* 1:19 21 (reprinted in 1993, *Rangelands* 15(3):109–110)
- McArthur E.D., Welch, B.L. 1982. Growth rate differences among big sagebrush (*Artemisia tridentata*) accessions and subspecies. *Journal of Range Management* 35: 396-401.
- McArthur, E.D., Stevens, R., Monsen S.B. 1995. Adaptation and success of big sagebrush and rubber rabbitbrush on disturbed sites. Pages 811-823 in G.E. Schuman and G.F. Vance, editors. *Proceedings – National Meeting of the American Society for Surface Mining and Reclamation*. Gillette, Wyoming.
- Meyer, S.E. 1990. Seed source differences in germination under snowpack in northern Utah. Pages 184-191 in 5th Billings Symp. on Disturbed Land Rehabilitation. Vol. I. Hardrock Waste, Analytical, and Revegetation. Montana State Univ. Reclamation Res. Unit Pub. 9003. Bozeman, MT.

- Meyer, S.E., Monson, S.B. 1990. Seed-source differences in initial establishment for big sagebrush and rubber rabbitbrush. Pages 200-208 in E.D. McArthur, E.M. Romney, S.D. Smith and P.T. Tueller, editors. Proceedings of a symposium on cheatgrass invasion, shrub die off and other aspects of shrub biology and management. Genral Technical Report INT-276, Intermountain Research Station, Ogden UT.
- Meyer, S.E., Monsen, S.B., McArthur, E.D. 1990. Germination response of *Artemisia tridentata* (Asteraceae) to light and chill: patterns of between population variation. Botanical Gazette 151(2):176–183.
- Miao, S.L., Carstenn, S. 2006. A new direction for large-scale experimental design and analysis. Frontiers in Ecology and the Environment 4(5): 227.
- Miller, R.F., Tausch, R.J. 2002. The role of fire in juniper and pinyon woodlands: a descriptive analysis. Pages 15-30 in K.E.M. Galley and T.P. Wilson, editors. Proceedings: The First National Congress on Fire, Ecology, Prevention, and Management. San Diego, CA, Nov. 27 - Dec. 1, 2000. Tall Timbers Research Station, Tallahassee, FL.
- Miller, R.F., Wigand, P.F. 1994. Holocene changes in semiarid pinyon-Juniper woodlands. BioScience 44(7):465–474.
- Miller, R.F., Bates, J.D., Svecar, T.J., Pierson, F.B., Eddleman, L.E. 2005. Biology, ecology, and management of western juniper. Technical Bulletin 152, Oregon State University, Agricultural Experiment Station. Corvallis.
- Monsen S.B. 1999. Establishment of Big Sagebrush (*Artemisia tridentata*) in Semiarid Environments Pages 81 -86 in P.G. Entwistle et al. editors. Proceeding: Sagebrush/Steppe Ecosystems. Bureau of Land Management Boise State University: Boise, ID.
- Moore, R.P. 1972. Tetrazolium staining for assessing seed quality. Pages 347-366 in W. Heydecker, Seed Ecology. Butterworths, London.
- Noss, R.F., Peters, R.L. 1995. Endangered ecosystems, a status report on America's vanishing habitat and wildlife. Defenders of Wildlife, Washington, D.C.
- Noss, R.F., LaRoe, E.T., Scott, J.M. 1995. Endangered Ecosystems of the United States: A Preliminary Assessment of Loss and Degradation, Biological Report 28, USDI National Biological Service, Washington, DC, 58 pp.
- Owens, M.K., Norton, B.E. 1992. Interactions of grazing and plant protection on basin big sagebrush (*Artemisia tridentata* ssp. *tridentata*) seedling survival. Journal of Range Management 45(3):257–262.
- Perryman B.L., Maier, A.M., Hild, A.L., Olson, R.A. 2000. Demographic characteristics of three *Artemisia tridentata* subspecies. Pages 320-329 in Wagner L editor Billings Land Reclamation Symposium 2000: Striving for Restoration, Fostering Technology and Policy for Reestablishing Ecological Function. Montana State University, Reclamation Research Unit: Billings, MT.
- Peters J. editor. 2000. Tetrazolium Testing Handbook 2000. Association of Official Seed Analysts (AOSA), Contribution No. 29.
- Rau, B. 2005. Soil Physical, Chemical and Understory Plant Nutritional Response to Pinyon-juniper Encroachment and Prescribed Fire in a Central Nevada Woodland Master's Thesis. University of Nevada, Reno.
- SAS Institute Inc. 1990. SAS/STAT ver. 9.1. SAS Institute Inc. Cary, NC.
- Suring, L.H., Wisdom, M.J., Tausch, R.J., Miller, R.F., Rowland, M.M., Schueck, L., Meinke, C.W. 2005. Modeling threats to sagebrush and other shrubland communities. Pages 114-149 in M.J. Wisdom, M.M. Rowland, and L.H. Suring, editors. Habitat threats in the sagebrush ecosystems: methods of regional assessment and applications in the Great Basin. Alliance Communications Group, Lawrence, KS.
- Tisdale, E.W., Hironaka, M. 1981. The sagebrush-grass region: A review of the ecological literature. Bulletin 33. Moscow, Idaho. Forestry, Wildlife and Range Experiment Station.
- Wisdom, M.J., Rowland, M.M., Suring, L.H. 2005. Habitat threats in the sagebrush ecosystem: methods of regional assessment and applications in the Great Basin. Alliance Communication Group, Lawrence Kansas.
- Wirth, T.A., Pyke, D.A. 2003. Restoring forbs for sage grouse habitat: fire, microsite establishment methods. Restoration Ecology 11(3):370-377.
- Wright, H.A., Klemmedson, J.O. 1965. Effect of fire on bunchgrasses of the sagebrush-grass region in southern Idaho. Ecology 46(5):680–688.
- Wroblewski, D.W., Kauffman, J.B. 2003. Initial effects of prescribed fire on morphology, abundance, and phenology of forbs in big sagebrush communities in southeastern Oregon. Restoration Ecology 11(1):82-90.
- Young, J.A., Evans, R.A. 1986. History of crested wheatgrass in the Intermountain area. Pages 21–25 in K.L. Johnson, editor. Crested wheatgrass: its values, problems, and myths. Utah State University, Logan, Utah.

Examining Endophyte Interactions within Fourwing Saltbush (*Atriplex canescens*)

Mary E. Lucero¹, Jerry R. Barrow¹, Isaac Reyes-Vera², and Ruth Sedillo¹

ABSTRACT

Cryptic endophyte consortia associated with embryonic tissues of fourwing saltbush (Atriplex canescens) are believed to serve mutualistic functions. However, the complexity of these endophyte communities makes cause-and-effect relationships difficult to establish. In fourwing saltbush, cryptic endophytes are thought to enhance drought and salt tolerance, nutrient acquisition, germination, and establishment. Some of these traits can be conferred to alternate host plants by co-culturing seedlings in vitro with micropropagated fourwing saltbush callus. Here we describe an evolving process for: 1) identifying arrays of cryptic, seed borne endophytes associated with fourwing saltbush; 2) evaluating early vigor of grasses and other plants endowed with fourwing saltbush endophytes; 3) selecting plant-endophyte combinations with increased tolerance to abiotic stressors, and 4) monitoring the persistence of transferred endophytes across plant generations in increasingly complex habitats. This process is designed to evaluate specific plant-fungal interactions that influence host tolerance to salinity, but may be broadly adaptable for selecting endophyte-enhanced plant germplasms with a variety of desired traits.

INTRODUCTION

The utility of fourwing saltbush and other *Atriplex* species for restoration, remediation, and forage combined with global distribution of the genus, halophyte physiology, and complex genetics have helped *Atriplex* to rank among the most widely studied native shrub species (Abeliovich and others 1994; Adair, Andrews and others 1992; Bu, Bai and others 2007; Cibils, Swift and others 1998; Glenn, Olsen and others 1994; Kadereit, Gotzek and others 2005; Karimi and Ungar 1989; Ortiz-Dorda, Martínez-Mora and others 2005; Ostyina, McKell and others 1984; Sanderson and McArthur 2004; Simon, Glenn, Pfister and others 1996; Ueckert and Petersen 1991). Yet many of the interesting traits attributed to *Atriplex* species, including drought and salt tolerance may be heavily influenced by internal fungal and bacterial endophytes (Barrow and Osuna 2002; Barrow, Osuna-Avila and others 2004). Experiments involving co-cultivation of endophyte-laden fourwing saltbush (*Atriplex canescens*) callus with non-host plants suggest cryptic endophytes associated with the plant can be transferred to diverse hosts, where they may increase vigor and establishment potential of the recipient plant (Barrow,

Lucero and others 2006; Lucero, Barrow and others 2006; Lucero, Barrow and others 2006; Lucero, Barrow and others 2008). Because the calli used in these studies was derived from embryonic tissue, the putatively transferred endophytes are considered capable of being carried to plant progeny within seeds. Thus, fourwing saltbush (and other *Atriplex* species) may serve as diverse warehouses of valuable endophytes which can be transferred to other species to improve recipient plant tolerance to arid rangeland environments. Unlike the more thoroughly described clavipitaceous endophytes that confer drought tolerance to cool season grasses, fourwing saltbush endophytes are not known to produce toxic alkaloids. Thus, they are deemed less likely than clavipitaceous endophytes to confer negative traits to foreign hosts. Their association with a plant species that thrives in harsh environments suggests significant potential for use in revegetation of arid lands.

A difficulty associated with the study of all microbial communities is that their cryptic, dynamic nature makes cause and effect relationships difficult to establish. Many endophyte species cannot be isolated or cultured independently of host plants (Barrow, Lucero and others 2008). In addition, even under micropropagation, incredible diversity exists within endophyte communities (Lucero, Barrow and others 2008) (Thomas, Swarna and others 2008; Thomas, Swarna and others 2008). Specific markers, such as the novel alkaloids indicative of certain clavipitaceous endophytes, have not been described for most endophyte species known to associate with fourwing saltbush, so detection to date has relied on universal molecular and histochemical methods.

Although these methods are helpful for illustrating the presence or absence of endophytes, they do little to distinguish between endophyte species. Thus, efforts to reproducibly detect and monitor transferred endophytes are prone to interpretation errors. For example, denaturing gradient gel electrophoresis (DGGE) techniques reveal DNA banding patterns in treated and untreated plants, supporting the hypothesis that endophytes had transferred from source callus to recipient plants (Lucero, Barrow and others 2006). However, subsequent isolation and sequencing of the co-migrating bands revealed significant differences between bands, with some sequences representing polymorphic plant DNA rather than transferred endophytes (unpublished). Tempered by these and other varied results, we have evolved a complex, cyclical process for examining endophyte interactions influencing fourwing

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Mary E. Lucero, Jerry R. Barrow, and Ruth Sedillo, USDA, Agricultural Research Service, Jornada Experimental Range, New Mexico State University

²Isaac Reyes-Vera, New Mexico State University.

saltbush biology (figure 1). In this proceedings we will summarize our process, describing various protocols we have developed or evaluated for 1) identifying arrays of cryptic, seed borne endophytes associated with fourwing saltbush 2) evaluating early vigor of grasses and other plants endowed with fourwing saltbush endophytes 3) selecting plant-endophyte combinations with increased tolerance to abiotic stressors, and 4) monitoring the persistence of transferred endophytes across plant generations in increasingly complex habitats. We will describe limitations to these techniques and emphasize the need for improved and complementary methods to monitor and evaluate endophytic microbes in complex habitats.

Identifying Arrays of Cryptic, Seed Borne Endophytes Associated With Fourwing Saltbush

Selection of Source Material

Endophyte communities associated with any plant material are remarkably complex, and may vary by both genotype

and habitat (Arnold and Lutzoni 2007; Porras-Alfaro, Herrera and others 2008). Hence, determination of endophytes associated with the species should include representative samples from populations that are both geographically diverse and are associated with varied habitats. We have identified potential sampling populations by utilizing Ecological Site Descriptions (ESD's) to create a spreadsheet database of sites in which *Atriplex* species are listed as part of the vegetative community. Sorting and filtering functions were used within the spreadsheet to identify ecological sites with varied soil and climate features. In this way, fourwing saltbush can be sampled from sites that vary in exposure to environmental stressors such as salinity and drought, and endophytes associated with each population can be evaluated. Comparative phylogenetic analyses and other correlations between endophytes, host plant genotypes, and habitats will be used to reveal endophyte species which are most correlated with host plants exposed to osmotically stressed habitats.

A process for examining endophyte interactions with fourwing saltbush

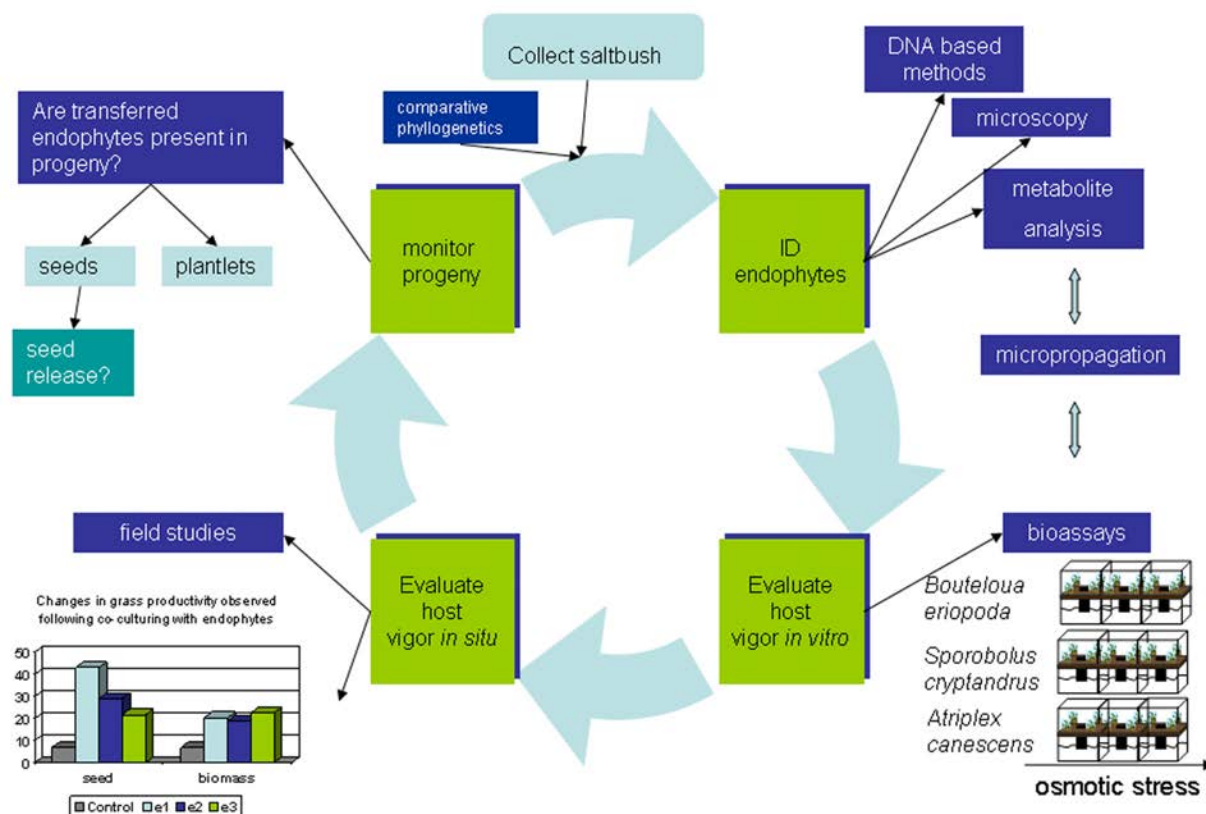


Figure 1—A multi-step process for examining interactions between various host plants and fourwing saltbush endophytes. Saltbush seeds harvested from well-described habitats provide starting material. Endophytes associated with these seeds are identified either as isolates or within aseptically germinated saltbush seedlings. Plant interactions with endophytes are first evaluated in vitro to minimize complexity of associated microbial populations. Shrubs and grasses inoculated with saltbush endophytes are exposed to osmotic stress, and endophytes that increase host stress tolerance are selected for greenhouse or field evaluations (in situ). Endophytes that improve host performance in field settings are examined for persistence in the progeny of inoculated plants.

Isolating Culturable Endophytes

Once source plants have been identified, harvested seeds from plants within each population serve as the source material for endophyte identification. Seeds are excised from the utricle and disinfested by soaking for a minimum of 30 minutes in a 1 percent solution of ZeroTol (BioSafe Systems, LLC.) in sterile water. Disinfested seeds are placed on petri-dishes containing malt extract agar, potato dextrose agar, or nutrient agar and incubated for 30 days. Endophytes detected growing on the agar are isolated, and DNA is extracted, amplified, cloned, and sequenced as previously described (Lucero, Barrow and others 2006). Additional aliquots of DNA are freeze dried for long term preservation so that additional gene regions can be amplified when necessary for more complete identification.

Isolating Unculturable Endophytes

Unculturable endophytes can be separated from the host plant by excising visible structures (usually hyphae) from plant tissues. Hyphae are often seen extruding from root surfaces of germinating seedlings, and are easily confused with root hairs. Sometimes, hyphae will also appear at the nodes between leaves and stems of micropropagated plants. Microdissection would be useful for extracting smaller fungal structures from plant tissues, such as hyphae surrounding stomata.

Optimizing Endophyte Detection

Fungal endophytes associated with fourwing saltbush can be detected using light microscopy with selective staining (Barrow and Aaltonen 2001; Barrow and Aaltonen 2003). However, this technique does not distinguish species, nor does it differentiate systemic fungi from those located only in a particular tissue, such as the roots. The addition of micropropagation to minimize interference from external microbes has provided an invaluable control, ensuring that only those microbes retained in embryonic tissues are detected (Barrow, Osuna-Avila and others 2004). Initiation of healthy micropropagated lines from embryos reduces the microbial community to highly endophytic species with potential for vertical transmission (Reyes-Vera 2008; Reyes-Vera, Potenza and others accepted).

Such cryptic endophyte communities are notoriously difficult to identify. Molecular techniques based on amplification and sequencing of conserved microbial gene sequences can provide a good starting point, but without specific primers, it is unlikely that all species present within a plant host will be detected (Lucero, Barrow and others 2008). Meanwhile, even within controlled, micropropagated lines, endophyte communities are complex and dynamic, such that species successfully detected at a given point in time may disappear from subsequent harvests. For this reason, newly established lines are considered to be more representative of natural populations, and combined detection and analysis techniques are most likely to provide informative results.

We continue to use PCR based methods in combination with micropropagation and microscopy to provide information about endophytes associated with fourwing saltbush. We are also exploring biochemical profiles of plants and isolated endophytes in the hopes of identifying chemical indicators of individual endophyte species.

Evaluating Early Vigor of Grasses and Other Plants Endowed With Fourwing Saltbush Endophytes

Micropropagation for Controlled Plant-Endophyte Interaction Bioassays

In addition to aiding identification of uncultured microbes, micropropagation has been valuable for exploring the physiological responses of plants to endophytes. Initial screening is accomplished by inoculating micropropagated plants with isolated endophytes, or with endophyte-laden callus (Lucero, Barrow and others 2006). These experiments have provided evidence that saltbush endophytes may improve establishment and reproductive potential of native grasses and of crop plants (Barrow and Lucero 2005; Barrow, Lucero and others 2006; Lucero, Barrow and others 2008). However, as noted above, detection and monitoring of mixed endophyte populations, even within micropropagated systems, remains problematic (Lucero, Barrow and others 2006; Lucero, Barrow and others 2008).

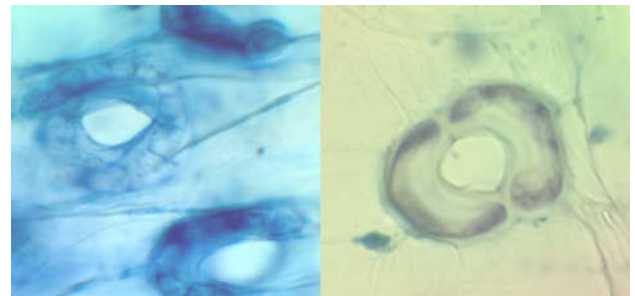


Figure 2—Leaf stoma of micropropagated *A. griffithsii* shoot cultures. The left image (untreated) contains blue staining fungal hyphae associated with guard cells of the stomatal complex. Treatment with the contact fungicide 26 GT® (Bayer Environmental Science) reduced endophyte colonization of stomata detectable with trypan blue (right image). 26 GT® contains the active ingredient iprodione (3-(3, 5-dichlorophenyl)-N-(1-methylethyl)-2, 4-dioxo-1-imidazolidinecarboxamide)), which inhibits the germination of fungal spores and mycelium.

Recently, treatment of micropropagated *A. griffithsii* shoot cultures with the contact fungicide 26 GT® (Bayer Environmental Science) has reduced endophyte colonization of stomata detectable with trypan blue (figure 2). 26 GT® contains the active ingredient iprodione (3-(3, 5-dichlorophenyl)-N-(1-methylethyl)-2, 4-dioxo-1-imidazolidinecarboxamide)), which inhibits the germination of fungal spores and mycelium. It is not yet clear whether

repeated doses will eliminate all trypan blue staining structures associated plant tissues, or how such treatments will affect culture viability. However, this is a positive step towards the development of an endophyte-reduced line, which would provide an invaluable control for examining plant interactions with various endophytes.

Meanwhile, micropropagated systems with mixed endophyte consortia remain useful, if not ideal, for examining endophyte influences on host plant physiology. Both host plant genetics and abiotic environments are tightly controlled, and microbial variability is minimized. Hence, the influence of various endophytes on plant performance can be evaluated by inoculating host and non-host plants with either isolated endophytes or uncultured endophyte consortia retained in saltbush callus.

Selecting Plant-Endophyte Combinations With Increased Tolerance to Abiotic Stressors

Salt Tolerance Bioassays

The ability of fourwing saltbush to tolerate arid environments is due in part to its halophytic traits. Hence, saltbush endophytes likely to influence host plant tolerance to salt may be identified by screening plant-endophyte combinations for increased tolerance to high concentrations of mineral salts. We perform salt tolerance bioassays in culture tubes containing Murashigie and Skoog's medium supplemented with 0 – 3 percent Instant Ocean®, a commercially prepared mixed salt (Murashige and Skoog 1962). Inoculated and uninoculated seedlings or micropropagated plantlets cultured at varied salt concentrations are compared by measuring root and shoot areas and whole plant biomass. Differences can often be detected within as little as two weeks growth, facilitating evaluation of multiple endophyte combinations. Plant-endophyte combinations that exhibit increased salt tolerance can be evaluated microscopically for changes in endophyte distribution patterns, bladder cell formation, stomatal density, and other variables important for osmoregulation.

Both saltbush and native grass hosts have been utilized as endophyte recipients in these bioassays. Advantages to using grasses as recipients in bioassays include rapid germination, small plant size, ease of micropropagation, and freedom from intrinsic fourwing saltbush endophytes. Grasses are also useful for evaluating the potential of saltbush endophytes in alternate hosts, where the ability to confer salt tolerance could be valuable not only for native plant restoration, but also for agricultural production or biofuel development.

Alternate Bioassays

Although our efforts are currently focused on assays to screen for endophytes that improve host plant productivity against varied salinity gradients. Since many of the same

biochemical pathways regulate other abiotic stressors, such as drought and heat stress responses, it is likely that selected plants will also exhibit tolerance to other arid land conditions. Hence, assays to specifically select for heat, drought, or cold tolerance are not being implemented at this time.

Instead, selected plants are monitored for differences in endophyte populations, and endophytes that appear to confer salt tolerance are monitored for persistence in plants as they are hardened and transferred from in-vitro to greenhouse propagation.

Monitoring the Persistence of Transferred Endophytes Across Plant Generations In Increasingly Complex Habitats

Co-cultivation of fourwing saltbush callus with native grasses has produced grass plants that exhibit high productivity in field trials (Lucero, Barrow and others 2008). However, questions regarding the nature of transferred endophytes and the degree to which they persist in novel hosts remain. Efforts to detect transferred endophytes with DNA analysis have provided variable results, suggesting transferred endophytes have either diminished over time, or have escaped detection in the presence of other microbes. Clearly, more robust monitoring methods are needed in order to evaluate the persistence of transferred microbes. Yet even methods that prove robust in laboratory trials may be prone to error in more complex greenhouse and field environments (Lueders and Friedrich 2003; Acinas, Sarma-Rupavtarm and others 2005). Therefore, validation of candidate markers for endophyte monitoring mandates availability of genetically similar control- and endophyte-enhanced plant materials cultivated in increasingly complex habitats.

To provide a series of varied, complex environments in which to evaluate the long term performance of grasses inoculated with endophytes from fourwing saltbush we are preparing a series of common garden plantings. Two black grama genotypes will be inoculated with saltbush endophytes as previously described (Lucero, Barrow and others 2008). DNA harvested from treated and untreated plants will be screened to evaluate changes in associated microbial communities. Replicate control and inoculated plants will be transplanted and maintained on three disturbed sites in southern New Mexico. Annual monitoring of transplants using a grid point-intercept method will be used to evaluate the establishment and productivity of treated and untreated plants. Seasonal changes in endophyte populations will be monitored using a combination of DNA based methods and microscopy. If planned biochemical analysis of saltbush endophyte communities reveals chemical indicators, such as alkaloids, that can be used to monitor endophyte presence, these will be included in subsequent monitoring.

CONCLUSION

Although endophytes appear universally associated with higher plants, the difficulties associated with detection and monitoring have largely limited studies of endophyte influences on plant communities to those endophyte species that can be monitored with specific indicators, including unique alkaloids or known molecular markers (Hahn, Nickel and others 1999) (Bacon, Porter and others 1977; Barrow, Lucero and others 2008). Nonetheless these limited studies have clearly demonstrated profound effects endophytes can have on larger ecosystems (Clay and Schardl 2002; Bultman and Bell 2003; Rudgers, Koslow and others 2004; Clay, Holah and others 2005).

A variety of dark septate and amorphic fungal endophytes have been observed in apparently mutualistic association with fourwing saltbush (Barrow, Havstad and others 1997; Barrow and Aaltonen 2001; Barrow and Aaltonen 2001; Barrow and Osuna 2002; Barrow 2003; Barrow, Osuna-Avila and others 2004). These endophytes are worthy of in depth study because their beneficial association with a hardy, arid land forage shrub and their potential to confer desirable traits to alternate hosts may provide a variety of useful applications for restoration and production agriculture (Lucero, Barrow and others 2006; Lucero, Barrow and others 2008). The complexity of the saltbush endophyte community mandates implementation of a complex, multifaceted system for identification and characterization of those endophytes which contribute to plant hardiness.

Our combined approach of comparing endophytes from diverse, well defined saltbush habitats to identify plant- and habitat-specific correlations; developing biomarkers for monitoring specific endophytes; evaluating endophyte interactions with host and non-host plants in highly controlled, in-vitro bioassays; and examining endophyte persistence in habitats of varied complexity is expected to reveal important plant-endophyte interactions that can be utilized to improve establishment and productivity of a variety of plant species in arid lands.

ACKNOWLEDGEMENTS

This work was supported by funds appropriated to the USDA Agricultural Research Service Jornada Experimental Range and an interagency agreement with the Bureau of Land Management Las Cruces Field Office. We thank Jaime Midez for assistance with in vitro and greenhouse bioassays, Jason Dunwell for maintenance of micropropagated plant lines.

REFERENCES

- Acinas, S.G., Sarma-Rupavtarm, R., Klepac-Ceraj, V., Polz, M.F. 2005. PCR-induced sequence artifacts and bias: Insights from comparison of two 16S rRNA clone libraries constructed from the same sample. *Applied and Environmental Microbiology*. 71(12):8966-8969.
- Adair, L.S., Andrews, D.L., Cairney, J., Funkhouser, E.A., Newton, R.J., Aldon, E.F. 1992. Characterizing gene responses to drought stress in fourwing saltbush *Atriplex canescens* (Pursh.) Nutt. *Journal of Range Management*. 45:454-461.
- Arnold, A.E., Lutzoni, F. 2007. Diversity and host range of foliar fungal endophytes: Are tropical leaves biodiversity hotspots? *Ecology*. 88(3): 541-549.
- Bacon, C.W., Porter, J.K., Robbins, J.D., Luttrell, E.S. 1977. *Epichloe typhina* from toxic tall fescue grasses. *Applied and Environmental Microbiology* 34(5):576-581.
- Barrow, J., Aaltonen, R. 2003. A staining method for systemic endophytic fungi in plants. *Emerging Concepts in Plant Health Management*. R.T. Lartey and Caesar A.J. Kerala, India, Research Signpost: 61-67.
- Barrow, J., Aaltonen, R.E. 2001. A method evaluating internal colonization of *Atriplex canescens* (Pursh) Nutt. roots by dark septate fungi and how they are influenced by physiological activity. *Mycorrhiza*. 11:199-205.
- Barrow, J.R. 2003. Atypical morphology of dark septate fungal root endophytes of *Bouteloua* in arid southwestern USA rangelands. *Mycorrhiza* 13(5):239.
- Barrow, J.R., Aaltonen, R.E. 2001. Evaluation of the internal colonization of *Atriplex canescens* (Pursh) Nutt. roots by dark septate fungi and the influence of host physiological activity. *Mycorrhiza* 11(4):199-205.
- Barrow, J.R., Havstad, K.M., Hubstenberger, J., McCaslin, B.D., Barrow, J.R. 1997. Seed-borne fungal endophytes on fourwing saltbush, *Atriplex canescens*. *Arid Soil Research and Rehabilitation* 11(4):307.
- Barrow, J.R., Lucero, M. 2005. Transfer and incorporation of heritable symbiotic fungi into non-host plants. Online: <http://patents.com/us-7901927.html>. Patent Application S.N. 11/500,702.
- Barrow, J.R., Lucero, M.E., and Reyes-Vera, I. 2006. Symbiotic fungi that influence vigor, biomass and reproductive potential of native bunch grasses for remediation of degraded semiarid rangelands. *Shrublands Under Fire: Disturbance and Recovery in a Changing World*. Fourteenth Wildland Shrub Symposium, Cedar City, Utah, USDA, Rocky Mountain Research Station. Proceedings RMRS-P-52.
- Barrow, J.R., Lucero, M.E., Reyes-Vera, I., Kavstad, K.M. 2008. Do symbiotic microbes have a role in plant evolution, performance and response to stress? *Communicative & Integrative Biology* 1(1):1-5.
- Barrow, J.R., Osuna-Avila, P., and Reyes-Vera, I. 2004. Fungal endophytes intrinsically associated with micropropagated plants regenerated from native *Bouteloua eriopoda* Torr. and *Atriplex canescens*. *In Vitro-Cellular & Developmental Biology-Plant* 40(6):608-612.
- Barrow, J.R., Osuna, P. 2002. Phosphorus solubilization and uptake by dark septate fungi in fourwing saltbush, *Atriplex canescens* (Pursh) Nutt. *Journal of Arid Environments* 51(3):449-459.
- Bu, Q., Bai, X., Wang, B. 2007. Accumulation and distribution of salt in leaves of *Atriplex triangularis* under salt stress. *Chinese Journal of Applied and Environmental Biology* 13(2):192.
- Bultman, T., Bel, G. 2003. Interaction between fungal endophytes and environmental stressors influences plant resistance to insects. *Oikos* 103(1):182-190.
- Cibils, A.F., Swift, D.M., McArthur, E.D. 1998. Plant Herbivore interactions in *Atriplex*: current state of knowledge. Rocky Mountain Research Station, Ogden, Utah, USDA Forest Service. RMRS-GTR-14.
- Clay, K., Holah, J., Rudgers, J.A. 2005. Herbivores cause a rapid increase in hereditary symbiosis and alter plant community composition. *Proceedings of the National Academy of Sciences of the United States of America* 102(35):12465-12470.
- Clay, K., Schardl, C. 2002. Evolutionary origins and ecological consequences of endophyte symbiosis with grasses. *American Naturalist* 160 (4 SUPPL.):S99 – S127.
- Glenn, E., Pfister, R., Brown, J.J., Thompson, T.L., O'Leary, J. 1996. Na and K accumulation and salt tolerance of *Atriplex canescens* (Chenopodiaceae) genotypes. *American Journal of Botany* 83(8):997.
- Glenn, E.P., Olsen, M., Frye, R., Moore, D., Miyamoto, S. 1994. How much sodium accumulation is necessary for salt tolerance in subspecies of halophyte *Atriplex canescens*? *Plant, Cell and Environment* 17(6):711.

- Hahn, D., Nickel, A., Dawson, J. 1999. Assessing *Frankia* populations in plants and soil using molecular methods. *FEMS Microbiology Ecology* 29(3):215-227.
- Kadereit, G., Gotzek, D., Jacobs, S., Freitag, H. 2005. Origin and age of Australian Chenopodiaceae. *Organisms Diversity and Evolution* 5(1):59.
- Karimi, S.H., Ungar, I.A., 1989. Development of epidermal salt hairs in *Atriplex triangularis* Willd. in response to salinity, light intensity, and aeration. *Botanical Gazette* 150(1):68-71.
- Lucero, M., Barrow, J., Osuna-Avila, P., Reyes-Vera, I., Duke, S. 2008. Enhancing native grass productivity by co-cultivating with endophyte-laden calli. *Rangeland Ecology and Management* 61:124-130.
- Lucero, M., Barrow, J.R., Sedillo, R., Osuna-Avila, P., Reyes-Vera, I. 2006. Genetic characterization of uncultured fungal endophytes from *Bouteloua eriopoda* and *Atriplex canescens*. *Shrublands Under Fire: Disturbance and Recovery in a Changing World*. Fourteenth Wildland Shrub Symposium, Cedar City, Utah, Rocky Mountain Research Station RMRS-P-52.
- Lucero, M.E., Barrow, J.R., Osuna-Avila, P., Reyes-Vera, I. 2008. A cryptic microbial community persists within micropropagated *Bouteloua eriopoda* (Torr.) Torr. cultures. *Plant Science* 174:570-575.
- Lucero, M.E., Barrow, J.R., Osuna-Avila, P., Reyes-Vera, I. 2006. Plant-fungal interactions in arid and semiarid ecosystems: large scale impacts from microscale processes. *Journal of Arid Environments* 65(2):276-284.
- Lueders, T., Friedrich, M. 2003. Evaluation of PCR amplification bias by terminal restriction fragment length polymorphism analysis of small-subunit rRNA and *mcrA* genes by using defined template mixtures of methanogenic pure cultures and soil DNA extracts. *Applied and Environmental Microbiology* 69(1):320-326.
- Murashige, T., Skoog, F., 1962. A revised medium for rapid growth and bioassays with tobacco tissue cultures. *Physiol Plant* 15:473-497.
- NRCS, Natural Resource Conservation Service. (12 November 2003). Ecological Site Information System, USDA.
- Ortiz-Dorda, J., Martínez-Mora, C., Correal, E., Simón, B., Cenis, J.L. 2005. Genetic structure of *Atriplex halimus* populations in the Mediterranean Basin. *Annals of Botany* 95(5): 827.
- Ostyina, R.M., McKell, C.M., Malecheck, J.M., Van Epps, G.A. 1984. Potential of Atriplex and other chenopod shrubs for increasing range productivity and fall and winter grazing use. General Technical Report – USDA, Forest Service (INT-172):215.
- Porras-Alfaro, A., Herrera, J., Sinsabaugh, R.L., Odenbach, K.J., Lowrey, T., Natvig, D.O. 2008. A novel root fungal consortium associated with a dominant desert grass. *Applied and Environmental Microbiology*: AEM.02769-07.
- Reyes-Vera, I. 2008. Dissertation. Plant and Environmental Sciences. Las Cruces, New Mexico State University.
- Reyes-Vera, I., Potenza, C., Barrow, J. 2008. Hyperhydricity reversal and clonal propagation of four-wing saltbush (*Atriplex canescens*, cultivated in vitro. *Australian Journal of Botany*. 56:358-362.
- Rudgers, J.A., Koslow, J.M., Clay, K. 2004. Endophytic fungi alter relationships between diversity and ecosystem properties. *Ecology Letters* 7(1):42-51.
- Sanderson, S.C., McArthur, E.D. 2004. Fourwing saltbush (*Atriplex canescens*) seed transfer zones. USDA, Forest Service, Rocky Mountain Research Station. General Technical Report RMRS-GTR-125.
- Simon, R.D., Abeliovich, A., Belkin, S. 1994. A novel terrestrial halophilic environment: The phylloplane of *Atriplex halimus*, a salt-excreting plant. *FEMS Microbiology Ecology* 14(2):99-109.
- Thomas, P., Swarna, G., Patil, P., Rawal, R. 2008. Ubiquitous presence of normally non-culturable endophytic bacteria in field shoot-tips of banana and their gradual activation to quiescent cultivable form in tissue cultures. *Plant Cell, Tissue and Organ Culture* 93(1):39-54.
- Thomas, P., Swarna, G., Roy, P., Patil, P. 2008. Identification of culturable and originally non-culturable endophytic bacteria isolated from shoot tip cultures of banana cv. Grand Naine. *Plant Cell, Tissue and Organ Culture* 93(1):55-63.
- Ueckert, D.N., Petersen, J.L. 1991. Selecting *Atriplex canescens* for greater tolerance to competition. *Journal of Range Management* 44(3):220.

Seeded Native Shrub Establishment on Disturbed Sites in Southwestern Wyoming

James S. Jacobs¹, Susan R. Winslow², Karen J. Clause¹, and Roger Hybner²

ABSTRACT

Critical wildlife habitat supporting mule deer, antelope, and sage grouse in high elevation rangeland and sagebrush ecosystems of southwest Wyoming is threatened by an expanding population and energy exploration and development. Our objective was to evaluate native shrub species establishment for restoration after disturbance. In October 2005, on a well-pad disturbance, 16 accessions of 12 native shrub species were drill-seeded in single species plots in a randomized complete block design with four replications. Also, two seed mixtures (Bridger and Shell) of grass, forb, and shrub species were broadcast- or drill-seeded, and one seed mixture (Shell) was hydro-seeded to separate areas outside of the replicated plots. Densities were sampled in September 2007. By 2007, nine of the 16 shrub accessions established in the replicated plots. *Atriplex aptera* had the greatest establishment at six plants/m² followed by *Atriplex canescens* at four plants/m². Establishment of the other seven accessions was similar and ranged from two to 0.4 plants/m². Where the Bridger mix was broadcast seeded, *Artemisia tridentata* ssp. *wyomingensis*, *Atriplex aptera*, and *Krascheninnikovia lanata* establishment was estimated at 10,000, 6,770, and 1,120 plants/ha, respectively. Where drill seeded, establishment of the three shrubs was 0, 4,480, and 2,240 plants/ha, respectively. Where the Shell mix was broadcast seeded, *Artemisia tridentata* ssp. *wyomingensis*, *Krascheninnikovia lanata*, and *Atriplex canescens* establishment was estimated at 21,300, 4,500, and 4,500 plants/ha, respectively. Where drill seeded, establishment for the three species was 7,800, 10,000, and 1,100 plants/ha, respectively. Where the Shell mix was hydro-seeded, only *Krascheninnikovia lanata* established at 1,100 plants/ha.

INTRODUCTION

For over 50 years, the Plant Materials Program of the Natural Resources Conservation Service (NRCS) has been assembling, testing, and releasing shrubby plant materials for conservation use. Since the release of *Lespedeza thunbergii* (Thunberg's Lespedeza) in 1952, the program has released 78 native subshrub and shrub accessions nationally for commercial propagation, and developed techniques for their successful use and management. On western wildlands, disturbance associated with resource extraction poses a conservation problem by threatening soil, water, wildlife, and plant resources. Developing

and evaluating plant materials for restoration of native plant communities on these sites has been a NRCS priority.

Shrubs accelerate soil development, enhance plant and animal diversity, increase forage production, and provide faunal cover and wildlife food (Booth 1985). Many species developed by the NRCS are important for restoration of western shrublands and wildlife habitat. *Artemisia ludoviciana* (white sagebrush) released in 1986, is used by small animals including sage grouse, rabbits, chipmunks, ground squirrels and other rodents, as well as antelope, mule deer, elk, mountain sheep and cattle (Stevens and others 2006). *Artemisia tridentata* ssp. *vaseyana* (mountain big sagebrush) released in 1987, provides thermal and security cover for wildlife, nesting cover for species of grouse and other upland birds, and winter forage for many large wildlife ungulates (Tilly and others 2006). Four species of *Atriplex* (saltbush) released from 1979 through 2002, are highly palatable to livestock and big game, provide diversity for reclamation projects, and make excellent screens, hedges, and barriers for erosion control (Ogle and St. John 2005). *Krascheninnikovia lanata* (winterfat) establishes fairly easily on disturbed sites, helps stabilize soils, and is considered good browse and forage for wildlife and livestock (Ogle and others 2003). *Purshia tridentata* (antelope bitterbrush) released in 1984 and 1997; provides erosion control on disturbed lands, is high quality spring and winter browse for wildlife, cover for small animals and birds, and the seeds are a food source for small animals (Dyer and Noller 2007).

Currently, the high demand for energy has resulted in an expansion of oil and gas extraction on the Pinedale Anticline and Jonah Gas Field regions of Sublette County, Wyoming. The high elevation sagebrush ecosystem of the area is important habitat to mule deer, antelope, sage grouse, and other wildlife. Restoration of native plant communities is important for maintaining wildlife habitat (Wood and others 1995) as well as preventing soil erosion (Grantz and others 1998) and noxious weed invasion. Challenges to restoration of native plant species in this ecosystem include short growing season, low and uncertain precipitation, high summer temperatures during drought periods, coarse soils with low water holding capacity, invasive plant species, and domestic and native ungulate herbivory. Loss of soil structure and compaction associated with drill site disturbance may impede restoration under these site conditions.

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹James S. Jacobs and Karen J. Clause, USDA Natural Resources Conservation Service, Bozeman, MT

²Susan R. Winslow and Roger Hybner, Bridger Plant Materials Center, USDA Natural Resources Conservation Service, Bridger, MT

Our objective was to evaluate cultivars, varieties, and germplasms of grass, forb, and shrub species for adaptation to the Pinedale Resource Area with emphasis on plant species native to the Rocky Mountain Region that provide forage production, a diverse ecosystem, and habitat for sage grouse, mule deer, antelope and other wildlife species, especially those dependent upon sagebrush communities. An additional objective was to evaluate seeding methods and seeding mixtures for adaptation and desired ecological diversity in the Pinedale Resource Area. In this proceeding, we report the results of native shrub establishment.

METHODS

Site Description

The project area is a 1.5 ha (3.8 ac) well-pad approximately 48 km (30 miles) south of Pinedale, WY (N ½ SW ¼ Section 10, T29N R107W). The nearly level site is within the cool central desertic basins and plateaus major land resource area (MLRA 34A) at an elevation of 2,193 m (7,195 ft). Average annual precipitation is 254 mm (10 inch), mainly in the form of snow. The peak growing season (70 days) is from May to July. The soils are mostly deep and well drained, fine sandy loam to sandy loam and loamy fine sand. Surface layers are 127 mm (5 inch) or more deep with sandy clay loam subsoils. The major soil series include Bluegrim-Forelle complex and Bluegrim-Cotha complex. The dominant vegetative cover type is classified as sagebrush steppe and the potential natural vegetation is estimated at 70 percent grass and grass-like plants, 10 percent forbs, and 20 percent woody plants. The key grass species are *Hesperostipa comata* (needle and thread), *Elymus lanceolatus* (thickspike wheatgrass), *Achnatherum hymenoides* (Indian ricegrass), and *Pseudoroegneria spicata* (bluebunch wheatgrass). The forbs include *Aster* spp. (aster), *Erigonum* spp. (buckwheat), *Dalea* spp. (clover), *Oenothera* spp. (evening primrose), *Erigeron* spp. (fleabane), and *Artemisia frigida* (fringed sagewort). Shrub cover is dominated by *Artemisia tridentata* (big sagebrush), *Ericameria nauseosa* and *Ericameria teretifolia* (rubber and green rabbitbrush), *Purshia tridentata* antelope bitterbrush, and *Krascheninnikovia lanata* (winterfat). The total annual production (air-dry weight) ranges from 786 kg/ha (700 lb/ac) in unfavorable years up to 1680 kg/ha (1,500 lb/ac) in favorable years.

Procedures

A 1.5 ha square oil/gas well pad was constructed in 2002. Approximately 15 cm (6 inch) of topsoil was stripped and stockpiled for 37 months. During reclamation following development, the topsoil was re-applied to the disturbed area. The soil was then ripped to mitigate heavy equipment compaction and restore infiltration and firmed and smoothed to create a seedbed so no more than a 1/8 inch imprint was left when walking over the site. Weeds were controlled using tillage and glyphosate at 3 kg active

ingredient/ha (2 qt product/ac) prior to planting. The entire 1.5 ha site was fenced with 1.8 m (6 ft) tall woven field fence.

In November 2005, 16 accessions of 12 shrub species (table 1) were seeded as monocultures using a four-row Kincaid Precision Cone-seeder in 1.2 by 6 m (4 by 20 ft) plots arranged in a randomized complete block study design with four blocks. The seeding rate was 2.15 million pure live seeds (PLS)/ha (871,000 PLS/ac). Two multi-species mixtures, Bridger and Shell (tables 2 and 3, respectively) were seeded on adjacent 0.17 ha (0.42 ac) plots. The 15-species Bridger mix was 40 percent grass, 33percent forb, and 27percent shrub was drill-seeded with a Truax® drill at a rate of 11.34 kg PLS/ha (10.12 lb PLS/ac) on one plot and broadcast using an ATV-mounted broadcast seeder at 22.68 kg PLS/ha (20.24 lb PLS/ac) on a separate plot. The 10-species Shell mix was 20 percent grass, 40 percent forb, and 40 percent shrub, and was drill-seeded with a Truax® drill at a rate of 5.8 kg PLS/ha (5.18 lb PLS/ac) on one plot and broadcast using an ATV-mounted broadcast seeder at 11.6 kg PLS/ha (10.34 lb PLS/ac) on a separate plot. In addition, one 0.4 ha (1 ac) plot was hydro-seeded with the Shell mixture at 11.6 kg PLS/ha (10.34 lb PLS/ac). The hydro-seeded area was prepared to a smooth slope without roughening to catch seed.

Table 1—The shrub species and their common names, cultivars, and accession numbers seeded on the Pinedale evaluation planting.

Scientific Name	Accession/	Common Name
<i>Artemisia frigida</i>	9087555	fringed sagewort
<i>Artemisia ludoviciana</i>	9087556	prairie sagewort
<i>Artemisia nova</i>	9087557	black sagebrush
<i>Artemisia tridentata</i>		
<i>spp. tridentata</i>	9087558	basin big sagebrush
<i>Artemisia tridentata</i>		
<i>spp. wyomingensis</i>	9087559	Wyoming big sagebrush
<i>Atriplex aptera</i>		Wytana fourwing saltbush
<i>Atriplex canescens</i>		Snake River Plains fourwing saltbush
<i>Atriplex confertifolia</i>	9087561	shadscale saltbush
<i>Atriplex falcate</i>		
(<i>A. gardnerii</i>)	9016134	Gardner's saltbush
<i>Atriplex tridentata</i>	9087560	basin saltbush
<i>Grayia spinosa</i>	9087563	spiny hopsage
<i>Krascheninnikovia</i>		
<i>lanata</i>		Northern Cold Desert winterfat
<i>Krascheninnikovia</i>		
<i>lanata</i>		Open Range winterfat
<i>Krascheninnikovia</i>		
<i>lanata</i>		Hatch winterfat
<i>Purshia tridentata</i>		Maybell antelope bitterbrush
<i>Purshia tridentata</i>	9087562	antelope bitterbrush

Sampling and Analysis

The plots were sampled in September 2007. In the replicated mono-species plots, emerged shrubs seeded in the plots were counted within 0.3 m (1 ft) linear sample plots at three randomly chosen locations in each of the middle two rows of the treatment plot for a total of six 0.3 m (1 ft) samples per plot. Plant height was measured from representative plants in each sample plot. Plant vigor and relative stand establishment were visually estimated for the entire plot. Additional comments were noted on items such as the presence of animal and bird life, and grazing or browsing activity. In the species mixture plots, plant density by species was counted in 10 randomly located 0.9 m² (9.62 ft²) sample plots. A relative rating of plant vigor and stand establishment, height, and other comments were also recorded.

Table 2—The species composition and percent of mixture of the Bridger Mixture at the Pinedale evaluation planting.

Scientific Name	Common Name	Percent of Mixture
<i>Elymus trachycaulus</i>	Pryor slender wheatgrass	12.8
<i>Elymus lanceolatus</i>	Critana thickspike wheatgrass	12.8
<i>Pascopyrum smithii</i>	Rosana western wheatgrass	12.8
<i>Poa secunda</i> (<i>P. sandbergii</i>)	High Plains Sandberg bluegrass	12.8
<i>Achnatherum hymenoides</i>	Rimrock Indian ricegrass	12.8
<i>Leymus cinereus</i>	Trailhead basin wildrye	7.7
<i>Achillea millefolium</i> var. <i>occidentalis</i>	Great Northern western yarrow	5.1
<i>Ratibida columnifera</i>	Stillwater prairie coneflower	5.1
<i>Linum lewisii</i>	Maple Grove prairie flax	5.1
<i>Phacelia hastata</i>	Silverleaf phacelia	5.1
<i>Sphaeralcea coccinea</i>	Scarlet globemallow	2.6
<i>Atriplex aptera</i>	Wytana fourwing saltbush	1.3
<i>Krascheninnikovia lanata</i>	Open Range winterfat	1.3
<i>Artemisia tridentata</i> ssp. <i>wyomingensis</i>	Wyoming big sage	1.3
<i>Artemisia frigida</i>	Fringed sage	1.3

Plant density and height data from the replicated plots were averaged for each treatment plot and the average was used in the analysis. Average density, height, vigor and stand estimate data from the replicated plots were analyzed using analysis of variance (Statistix 2003). The model included

replication and seeded species. Means were compared using Fisher's protected least significant differences test at the 0.05 level of confidence (Peterson 1985). Means were calculated for the density of shrub species counted in the species mixture plots.

RESULTS AND DISCUSSION

The analysis of variance showed the accessions, cultivars, and shrub species established differently in the replicated plots on the reclaimed well-pad site ($P \leq 0.05$). Two years after seeding, 'Wytana' *Atriplex aptera* density averaged six plants/m², which was greatest of all species (table 4). It also was one of the tallest, most vigorous, and had the best stand rating of the species (table 4). Snake River Plains *Atriplex canescens* density averaged four plants/m², not statistically different than 'Northern Cold Desert' *Krascheninnikovia lanata* (1.8 plants/m²), Open Range *K. lanata* (1.4 plants/m²), or the accession of *Artemisia nova* (black sage, 1.4 plants/m²). These densities meet the requirements of one shrub/m² on 20 percent of the area of revegetated new coal mines in Wyoming (Booth and others 1999). The densities of the later three shrubs were not statistically different than densities of *Artemisia tridentata* ssp. *tridentata*, 'Hatch' *K. lanata*, *Atriplex falcata*, or *Atriplex tridentata* (0.8, 0.8, 0.4, 0.4 plants/m², respectively). While these species densities were lower than other species, their vigor rated as well as species with greater density. *Artemisia frigida*, *Artemisia ludoviciana*, *A. tridentata* ssp. *wyomingensis*, *Atriplex confertifolia*, *Grayia spinosa*, and the two accessions of *Purshia tridentata* were not found in any of the sample plots. Evidence of browse was noted on *K. lanata*. Shrub densities on ten year old reclaimed coal-mined sites ranged from 0.09 to 1.92 plants/m² (Booth and others 1999).

Table 3—The species composition and percent of mixture of the Shell mixture at the Pinedale evaluation planting.

Scientific Name	Common Name	Percent of Mixture
<i>Artemisia tridentata</i> ssp. <i>wyomingensis</i>	Wyoming big sagebrush	41.50
<i>Poa secunda</i> (<i>P. sandbergii</i>)	Sandberg bluegrass	30.71
<i>Artemisia frigida</i>	Fringed sagewort	7.53
<i>Penstemon rydbergii</i>	Rydberg's penstemon	7.30
<i>Achnatherum hymenoides</i>	Indian ricegrass	4.68
<i>Achillea millefolium</i>	Native yarrow	4.60
<i>Krascheninnikovia lanata</i>	Winterfat – bearded	1.88
<i>Atriplex canescens</i>	Fourwing saltbush	0.86
<i>Sphaeralcea coccinea</i>	Scarlet globemallow	0.50
<i>Lupinus argenteus</i>	Silvery lupine	0.43

Where the Bridger species mix was drill-seeded, Wytana *Atriplex aptera* and Open Range *Krascheninnikovia lanata* densities averaged 4,480 and 2,240 plants/ha, respectively, in the 2007 sample plots. Where broadcast-seeded, *Artemisia tridentata* ssp. *wyomingensis*, Wytana *A. aptera*, and Open Range *K. lanata* densities averaged 10,000, 6,770, and 1,120 plants/ha, respectively. Where the Shell species mixture was drill-seeded, *A. tridentata* ssp. *wyomingensis*, *K. lanata*, and *Atriplex canescens* densities averaged 7,800, 10,000, and 1,100 plants/ha, respectively. Where the Shell species mixture was broadcast-seeded their densities averaged 21,300, 4,500, and 4,500 plants/ha, respectively. Only *K. lanata* established at 1,100 plants/ha where the Shell mix was hydro-seeded.

Table 4—Mean values generated from ANOVA for density, height, vigor, and relative stand establishment of 16 shrubs seeded in replicated plots at the Shell Field Evaluation Planting near Pinedale, Wyoming. Vigor and stand were rated on a scale of 1 to 10 with being the best. Means followed by the same letter are not significantly different determined by LSD ($P < 0.05$) from other means within the column.

Scientific Name	Density Per m ²	Height (cm)	Vigor	Stand
<i>Atriplex aptera</i>	6.2 a	18 ab	4.2 ab	5.5 b
<i>Atriplex canescens</i>	3.7 b	24 a	2.8 b	5.0 b
<i>Krascheninnikovia</i>				
<i>lanata</i> (Northern Cold Desert)	1.8 bc	12 bc	5.0 ab	9.0 a
<i>Krascheninnikovia</i>				
<i>lanata</i> (Open Range)	1.4 bc	10 bc	4.5 ab	7.8 a
<i>Artemisia nova</i>	1.4 bc	8 cd	6.3 a	8.8 a
<i>Artemisia tridentata</i>				
ssp. <i>tridentata</i>	0.9 c	17 ab	4.8 ab	8.5 a
<i>Krascheninnikovia</i>				
<i>lanata</i> (Hatch)	0.9 c	10 bc	5.0 ab	8.0 a
<i>Atriplex falcata</i>				
(<i>A. gardnerii</i>)	0.4 c	13 bc	4.3 ab	8.0 a
<i>Atriplex tridentata</i>	0.4 c	7 c	6.5 a	9.0 a
<i>Artemisia frigida</i>	0	0	0	0
<i>Artemisia ludoviciana</i>	0	0	0	0
<i>Artemisia tridentata</i>				
ssp. <i>wyomingensis</i>	0	0	0	0
<i>Atriplex confertifolia</i>	0	0	0	0
<i>Grayia spinosa</i>	0	0	0	0
<i>Purshia tridentata</i>	0	0	0	0
<i>Purshia tridentata</i>	0	0	0	0

The seed mixture plots demonstrate either seed mix applied at any of the three seeding methods can meet the post-mining shrub density requirement on coal-mined lands in Wyoming of one shrub/m² on 20 percent of the affected area (Booth and others 1999). The hydro-seeding

application of the Shell mixture resulted in the lowest shrub density and diversity, and lowest overall plant diversity (data not shown) of the five applications. Either drill- or broadcast-seeding the Shell mixture with a greater proportion of shrub seeds than the Bridger mixture resulted in greater than one shrub/m² and within the range of densities found in southwest Wyoming (Booth and others 1999). Broadcasting the Bridger mixture also resulted in shrub densities within this range. Broadcast-seeding favored the establishment of *Artemisia tridentata* ssp. *tridentata* regardless of the seed mixture. Newman and Redente (2001) found native plant communities seeded and established on a disturbed sagebrush site in northwestern Colorado remained persistent over 20 years, and inhibited colonization of neighboring species. Although irrigation was not a treatment in this study, initial irrigation increased the long-term production of the native plant community, especially the forbs and shrubs, on the Colorado site (Newman and Redente 2001).

Atriplex canescens may be the most seeded western shrub species on disturbed sites and its successful establishment has been widespread (Booth 1985). It is considered one of the more valuable shrubs for all classes of livestock and wildlife. It may aid the reestablishment of sagebrush by providing favorable microclimates for seedlings and supporting mycorrhizae. As a pioneer species, it is considered to facilitate succession in sagebrush communities. It has been suggested that successful *A. canescens* establishment may exclude other shrubs. However, on a revegetated coal-mined site in eastern Wyoming, seedling Wytana *Atriplex aptera* at densities ≤ 5 seedlings/m² were unlikely to deter the development of a sagebrush stand (Booth 2002).

ACKNOWLEDGEMENTS

The authors thank the Aimee Davison, biologist for the Shell Exploration and Production Company, Sublette County Conservation District, Bureau of Land Management, Wyoming Game and Fish Department for their help and cooperation with this project.

REFERENCES

- Booth, D.T. 1985. The role of fourwing saltbush in mined land reclamation: A viewpoint. *Journal of Range Management* 38: 562-565.
- Booth, D.T. 2002. Seed longevity and seeding strategies affect sagebrush revegetation. *Journal of Range Management*. 55:188-193.
- Booth, D.T., Gores, J.K., Schuman, G.E., Olson, R.A. 1999. Shrub densities on Pre-1985 reclaimed mine lands. *Restoration Ecology* 7: 24-32.
- Dyer, D., Noller, G.L. 2007. Plant Guide: Antelope bitterbrush *Purshia tridentata* (Pursh) D.C. USDA – NRCS Plants Database [Online] Available: <http://Plants.usda.gov> [August 14, 2007].

- Grantz, D.A., Vaughn, D.L., Farber, R., Kim, D., Zeldin, M., VanCuren, T., Campbell, R. 1998. Seeding native plants to restore desert farmland and mitigate fugitive dust and PM10. *Journal of Environmental Quality*. 27: 1209-1218.
- Newman, G.J., Redente, E.F. 2001. Long-term plant community development as influenced by revegetation techniques. *Journal of Range Management* 54: 717-724.
- Ogle, D.G., St John, L., Holzworth, L., Winslow, S.R. 2003. Plant Guide: Winterfat *Krascheninnikovia lanata* (Pursh) A.D.J. Meeuse & Smit. USDA – NRCS Plants Database [Online] Available: <http://Plants.usda.gov> [February 11, 2005].
- Ogle, D., St. John, L. 2005 Plant Guide: Fourwing saltbush *Atriplex canescens* (Pursh) Nutt. USDA – NRCS Plants Database [Online] Available: <http://Plants.usda.gov> [March 14, 2005].
- Peterson R.G. 1985. Design and analysis of experiments. New York (NY): Marcel Dekker Inc. 401 p.
- Statistix. 2003. Statistix 8 Analytical Software. Tallahassee, FL. 396 p.
- Stevens, M., Roberts, W., Anderson, M.K. 2006. Plant Guide: White sage *Artemisia ludoviciana* Nutt. USDA – NRCS Plants Database [Online] Available: <http://Plants.usda.gov> [January 4, 2006].
- Tilly, D.J., Ogle, D., St. John, L., Benson, B. 2006. Plant Guide: Big sagebrush *Artemisia tridentata* Nuttall. USDA – NRCS Plants Database [Online] Available: <http://Plants.usda.gov> [May 30, 2006].
- Wood, M.K., Buchanan, B.A., Skeet, W. 1995. Shrub preference and utilization by big game on New Mexico reclaimed mine land. *Journal of Range Management*. 48:431-437

Incorporating Historical Ecosystem Diversity into Conservation Planning Efforts in Grass and Shrub Ecosystems

Amy C. Ganguli¹, Jonathan B. Haufler¹, Carolyn A. Mehl¹, and Jimmie D. Chew²

ABSTRACT

Understanding historical ecosystem diversity and wildlife habitat quality can provide a useful reference for managing and restoring rangeland ecosystems. We characterized historical ecosystem diversity using available empirical data, expert opinion, and the spatially explicit vegetation dynamics model SIMPPLLE (SIMulating Vegetative Patterns and Processes at Landscape ScaLEs) for a landscape of approximately 946,000 ac in eastern Wyoming. We used SIMPPLLE to simulate plant community dynamics as a result of historical disturbance events (for example, fire, bison grazing, and prairie dog activity), climate, and landscape elements (for example, ecological site, proximity to water, and elevation) and their interactions to derive estimates of the historical range of variability for each grass/shrub ecosystem. For each NRCS designated ecological site we defined the historical states that occurred in the presence of grazing by native herbivores and fire, and identified the processes for movements among states within each site. For each historically occurring state within the delineated landscape we determined the mean ac it occupied and the range of variability (in other words, minimum and maximum ac a state occupied). Comparisons of historical grass/shrub ecosystem diversity of the area with existing conditions indicate that there have been significant changes, most notably the lack of representation of ecosystems dominated by grass species that typically decrease with grazing, the widespread presence of introduced species especially annual brome, and alterations to fire regimes. Through comparisons of historical ecosystem diversity with existing conditions, we identified specific plant communities that are underrepresented and in need of restoration to maintain ecosystem diversity and wildlife habitat.

INTRODUCTION

It has been suggested that loss of evolutionary habitats, or the environments in which species evolve, is one of the greatest threats to biological diversity (Templeton and others 2001). In an effort to address this threat many conservation planning initiatives have sought to identify historical reference conditions of ecosystems for assessment of existing conditions and identification of management goals or conservation priorities (Forbis and others 2007; Haufler and others 1996). Historical reference approaches

represent one type of coarse filter strategy for biological diversity conservation (Haufler 1999). This approach is based on the premise that the ecosystem diversity that occurred in an area over the past several hundred to thousand years defined biodiversity at the ecosystem and landscape levels, and provided the habitat that supported the species and genetic diversity of a landscape (Haufler and others 2002; Poiani and others 2000). This coarse filter approach has the primary objective of providing adequate ecological representation of all historically occurring ecosystems especially in the context of providing adequate wildlife habitat for species of concern.

Historical references describe ecosystem conditions that resulted from natural disturbance (in other words, fire, grazing, etc.) and human-influenced disturbance (in other words, native American), which created the dynamic conditions that supported the native species of an area prior to the influences of European settlement. Historical references are usually confined to a period less than 1000 years prior to European settlement, as these reflect the habitat conditions most relevant to the species that are present today (Morgan and others 1994). Historical disturbance regimes are the patterns of frequency and intensity that can be quantified using ecological evidence. Historical reference conditions are typically characterized using the Historical Range of Variability (HRV), which emphasizes that many ecosystems varied in amount, composition, and structure due to historical variations in climate and stochastic events (Aplet and Keeton 1999).

The most common approach used to characterize historical reference conditions are field assessments that quantify biological attributes of sites identified as being relatively free from anthropogenic alterations and where the ecological processes that a site evolved with are relatively intact (Stoddard and others 2006; USDA/NRCS 2003). Although this approach to characterizing historical reference conditions provides very useful information, appropriate reference sites are often lacking because of historical disturbance regime modification and land conversions (Nilsson and others 2007). Furthermore, relict reference sites are typically found in inaccessible areas characterized by rugged terrain, rocky outcrops, and areas of shallow soil depth (Shinneman and others 2008), which are rarely characteristic of the type of sites in the surrounding landscape. These types of reference sites also

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Amy C. Ganguli, Jonathan B. Haufler, Carolyn A. Mehl, The Ecosystem Management Research Institute, Seeley Lake, MT

²Jimmie D. Chew, USDA Forest Service, Rocky Mountain Research Station, Missoula, MT

have higher probabilities of having site conditions indicative of late historical states because of their protected landscape position.

Lack of appropriate reference areas are especially a problem in many rangeland systems where modern grazing practices, fire management policies, and wildlife “pest control” policies have resulted in very different ecosystem dynamics. Modern livestock operations often control the timing, duration, and level of utilization by livestock and most operations include infrastructure developments like water, fencing, and attractants to promote even utilization and homogeneous structure (Fuhlendorf and Engle 2001). In many areas of the Great Plains, fire suppression activities and land use practices that have reduced the probability of fire spread have drastically decreased the occurrence of fire compared to historical fire regimes resulting in modification of ecosystem structure and function (Bragg and Hurlbert 1976; Hoch and others 2002; Sieg and others 1999). Pest control (for example, prairie dog poisoning) has also had an impact on the amounts of this historical disturbance (Miller and others 1994). As an alternative to field assessments, simulation modeling efforts have been used to develop reference conditions in forested systems (Barrett 2001; Chew and others 2004), and LANDFIRE model components have been used to provide reference conditions in grass and shrub ecosystems in Nevada (Forbis and others 2007).

Most modern restoration efforts do not attempt to return landscapes to conditions that occurred prior to the influences of European settlement. Non-indigenous species and anthropogenically modified disturbance regimes characterize most modern landscapes and complicate restoration efforts. However, providing an understanding of the ecosystem diversity that occurred during an identified timeframe prior to European settlement provides critical reference information for defining and quantifying a baseline of what should be considered “natural” for an area. Below we describe the primary disturbances that historically influenced ecosystems within the Thunder Basin region and we describe the historical ecosystem diversity, including an estimate of the historical range of variability of ecosystems within a delineated, 946,000 ac area in Converse, Campbell, Weston, and Niobrara Counties in eastern Wyoming.

Review of Evolutionary Disturbance Mechanisms in Eastern Wyoming

Prior to Euro-American settlement, natural disturbance processes such as fire and grazing were primary influences on the ecosystem diversity that occurred in eastern Wyoming (Knight 1994). Additional disturbances included herbivory and burrowing activities of prairie dogs (Kotliar and others 1999). Native Americans interacted with and influenced ecosystem diversity for thousands of years, but typically in ways that used naturally occurring disturbance

processes to benefit their subsistence strategies, such as using fire to create better wildlife habitat for hunted species or maintaining travel corridors in more open conditions (Williams 2005).

Fire

As in much of the Great Plains grasslands, fire was a relatively common disturbance event in eastern Wyoming prior to European settlement (Daubenmire 1968; Fisher and others 1987; Hann 2003; Perryman 1996; Perryman and Laycock 2000). Historically, fires were started by lightning (Higgins 1984; Komarek 1964) and the activities of Native Americans (Higgins 1984; Williams 2005). Native Americans used fire throughout the year. In contrast, fires started by lightning occurred mostly in hot and dry summer conditions, late growing season, or during the dormant season (Komarek 1964; Perryman and Laycock 2000). Within fire maintained landscapes, microhabitats that prevent or slow fire spread existed in riparian zones, badlands, ravines, and other fire-protected locations like prairie dog colonies (Anderson 1990). Furthermore, fire return intervals were also influenced by climate and by previous grazing disturbance. Fire effects on grassland ecosystems are a function of fire frequency, intensity, and timing, as well as the interaction of these factors with herbivory (Engle and Bidwell 2001). Fire can influence grassland vegetation in a number of ways including changes to productivity, composition, and structure (Engle and Bidwell 2001).

Bison Grazing

Great Plains grasslands were grazed by many herbivores, however, no species was more influential than bison in shaping the grassland ecosystems of Wyoming (Knapp and others 1999). Intensity of bison grazing was influenced by actively growing plants (McNaughton 1985), juxtaposition to water sources and recent fire events. Bison, like most herbivores, require a regular supply of water. Sites surrounding rivers, lakes, and ponds typically receive a disproportionate amount of heavy grazing. In contrast, sites farthest from water sources typically receive lower levels of grazing (Soper 1941). Many researchers have found that a recently burned site will attract bison (Bamforth 1987; Biondini and others 1999; Coppedge and Shaw 1998; Frank and others 1998). The release of soil nutrients and the corresponding rapid new growth represents high-quality forage for several seasons following a fire event. At the landscape level, historical fire and grazing disturbance regimes interacted to provide a mosaic of structural and successional conditions across grassland ecosystems (Fuhlendorf and Engle 2001). Fire and bison grazing interacted because recently burned sites attracted bison grazing, and subsequently the amount of forage removed from a site and its distribution in the landscape determined the probability and intensity of the next fire event (Fuhlendorf and Engle 2001). Thus, the combination of fire

and grazing yielded the dynamic habitat mosaic and landscape heterogeneity to which prairie wildlife species were well adapted (Fuhlendorf and Engle 2001; Fuhlendorf and Engle 2004).

Black-tailed Prairie Dogs

Prior to European settlement, black-tailed prairie dogs occurred extensively throughout eastern Wyoming (Wuerthner 1997). Black-tailed prairie dogs are considered an historical disturbance component in Wyoming due to the effect of their colonies on grassland ecosystems. Prairie dogs alter prairie ecosystems by creating above and below ground disturbances that produce large and unique habitat patches and thereby alter the structural and functional properties of grassland ecosystems (Whicker and Detling 1988).

METHODS

Description of Historical States

We described all the terrestrial ecosystems or “states” that historically occurred (in other words, the ecosystem diversity) in Thunder Basin as a function of the different abiotic factors (for example, soils, climate, etc.) and disturbances that historically occurred (for example, fire, grazing, prairie dog activities) (Haufler and others 1996; Haufler 2000). We utilized the Natural Resources Conservation Service’s (NRCS) ecological site classification system that uses soils as the basic mapping unit (USDA/NRCS 2003). Ecological sites provide valuable information on abiotic conditions and they provide information on the potential states that could occur on each site. This information must then be further developed to quantify the actual historical states and their amounts that occur across any specific ecological sites. Thus, we included fire and large herbivore grazing (in other words, bison) as the primary disturbance mechanisms that historically operated in grass and shrub ecosystems of eastern Wyoming and described the historical states occurring on each ecological site (see figure 1 for an example of the descriptions of historical states that occurred as a function of historical disturbance regimes). We predicted fire and grazing disturbance transitions for each ecological site using the best available information on ecosystem and plant species response. For example, we used canopy cover of plant species that typically increase or decrease with different levels of grazing pressure as indicators of historical states driven by different grazing regimes. We identified plant species likely to occur in a particular ecological site and how those species typically respond to grazing from NRCS developed ecological site descriptions, as well as, input from a team of range ecologists knowledgeable of the plant dynamics of the area. We used canopy cover of big sagebrush (*Artemisia tridentata*) as an indicator of historical states driven by short or long interval fire regimes, where some states were

classified as being characteristic of long fire return intervals if they had >10 percent big sagebrush cover. We characterized the historical fire regime for this area by using information developed for the fire regime condition class Interagency Handbook Reference Conditions (Hann 2003), as well as, supplemental literature (Perryman 1996; Perryman and Laycock 2000). We estimated plant dynamics and rates of change for each plant species included as indicators of either fire return interval or grazing level based on input from the team of rangeland ecologists and species information.

Determining the Historical Reference: Modeling Historical Ecosystem Diversity

We modeled historical grass/shrub vegetation dynamics in Thunder Basin using a spatially explicit landscape model SIMPPLLE (SIMulating Patterns and Processes at Landscape scaLEs) (Chew and others 2004). We used SIMPPLLE to simulate plant community dynamics as a result of natural disturbance events (for example, fire, bison grazing, and prairie dog activity), climate, and landscape elements (for example, ecological site, proximity to water, and elevation) in the delineated planning area in eastern Wyoming. Although SIMPPLLE has a variety of potential applications, we specifically used SIMPPLLE to derive the historical range of variability (HRV) for each grass/shrub ecosystem and we represented the HRV as the average, minimum, and maximum number of ac that each historical state occupied in our simulations.

Within the planning area we divided the landscape into 10 ac cells and assigned each cell as a specific vegetation unit defined by its ecological site and by its historical state which was based on its vegetation composition. We identified starting points (in other words, historical states based on plant species composition) for each 10 ac cell using generalized vegetation descriptions included in historical public land survey records (PLSRs) conducted in the 1880s. The 1880 PLSR information provided general descriptions of the dominant plant types (for example bunchgrass, sagebrush, etc.) and was prone to surveyor bias, however, this information offered at least a generalized starting point for the modeling effort that was based on spatially reported data from the 1880s.

Landscape features that were static components in each simulated area included ecological sites, aquatic areas, riparian areas, estimated locations of historical prairie dog colonies, and elevation. To map ecological sites for the planning area, which occurred within the Northern Rolling High Plains Major Land Resource Area (MLRA 58B), we used the NRCS ecological site classification system that uses soils as the basic mapping unit (USDA/NRCS 2003). We used National Hydrography Data and NRCS ecological sites to map wetland, riparian, and aquatic areas and digital elevation models to map elevation within the project area.

The approximate locations of historical prairie dog colonies were derived from a prairie dog suitability assessment conducted for this area that was based on preferred soils, other terrain features, and knowledge of existing prairie dog colonies.

The response of key plant species to climate (in other words, precipitation) and disturbance (in other words, fire and grazing) were tracked annually in SIMPPLLE for each 10 ac cell. Within a given year plant species within each cell were subject to change based on climate (for example, above average, average, or below average precipitation), grazing (for example, light, moderate, or heavy grazing), and the occurrence of fire. Subsequently each 10 ac cell was given an ecosystem classification that placed it into a historical state within each NRCS designated ecological site based on its species composition. That is, classification rules were developed that used percent cover of species within a cell to identify what historical state it belonged to, and over time climate and disturbance induced changes in plant species composition caused shifts among historical states. We based the plant species response parameters to climate and disturbance, as well as shifts among historical states on expert opinion from a team of rangeland ecologists, and on scientific literature.

Fire starts were caused by lightning strikes in this model and were stochastically selected, resulting in variations designed to simulate historical variations in lightning

caused fires over time. The number of lightning strikes was adjusted in the model to cause increases or decreases in the number of fire starts, but the overall influence of fire was more dependent on the burn patterns than on the number of fire starts. Once a fire started in a given cell it had the opportunity to spread to adjacent cells until it encountered cells that reduced the ability of fire to spread (see below), or encountered a stochastic weather ending event. The probability of fire occurrence was influenced by the climate (precipitation and temperature) in a given year and the grazing history on individual units (for example, a heavily grazed 10-ac unit in a given year had a lower probability of burning, whereas a lightly grazed 10-ac unit had a higher probability of burning). Fire spread probabilities were also influenced by fixed landscape features, such as prairie dog colonies, gullied land, and aquatic/riparian areas that provided natural fire breaks. Terrestrial ecological sites that had low probability of fire because of sparse fine fuel included the following ecological sites; saline upland, very shallow, badlands, and gullied land which combined represented 84,253 ac within the planning area. The remaining terrestrial ecological sites clayey, shallow clayey, loamy, shallow loamy, sands, sandy, shallow sandy, and shallow hilly representing 810,978 ac, had a fire return probability of 11 years that ranged from approximately 3 to 15 years, with some areas exhibiting low fuel levels burning infrequently or never.

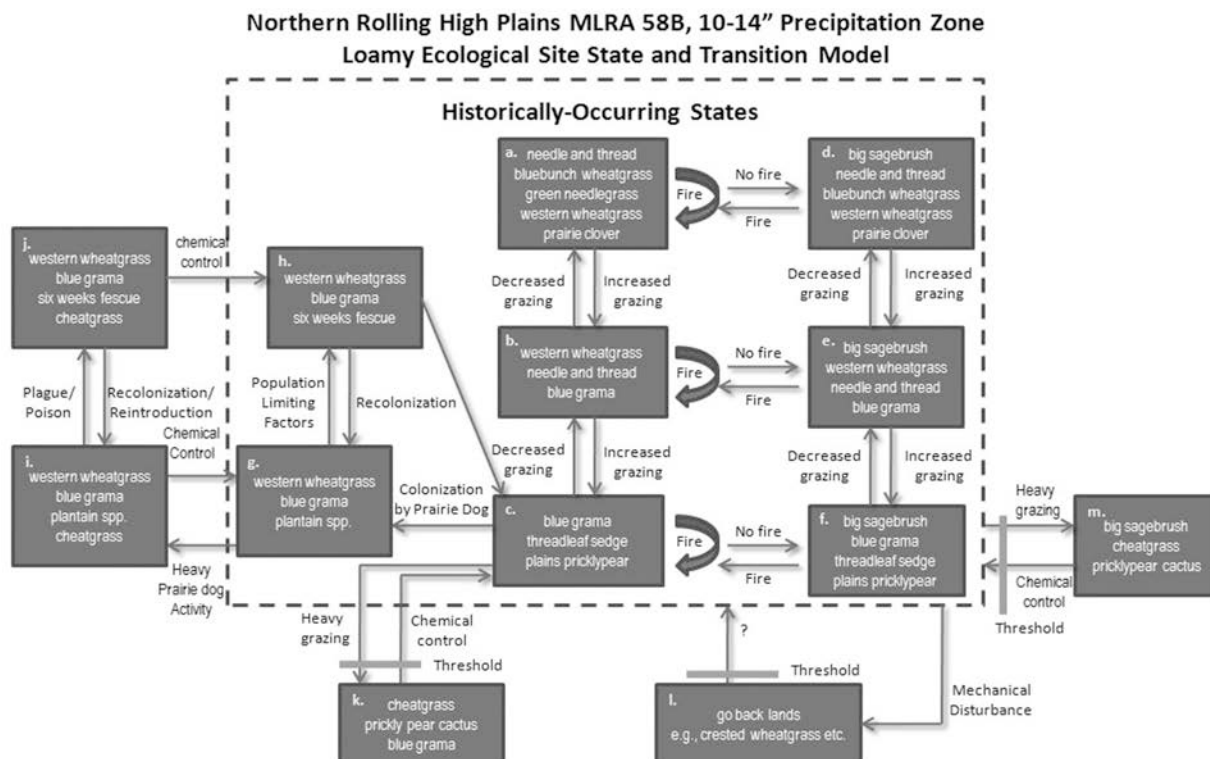


Figure 1—Expanded state and transition model highlighting the states, disturbance pathways, and transitions that historically occurred on loamy ecological sites within the Northern Rolling High Plains Major Land Resource Area (MLRA 58B).

Bison grazing intensity was dependent on the proximity of the 10-ac vegetation units to water and the fire history of the vegetation units within the areas simulated. For instance, based on knowledge of bison grazing behavior it was assumed that the closer the 10-ac vegetation units were to water and the more recently burned the vegetation units were, the heavier bison would graze. Vegetation units located between 0 to 5,280 feet away from water had a higher probability of receiving heavy bison grazing, whereas vegetation units located between 5,281 to 15,840 and 15,841 or greater feet away from water had increasingly higher probabilities of receiving moderate or light grazing, respectively. Likewise, the probability of heavy grazing on 10-ac vegetation units 1 to 2 years after a fire was higher, whereas 3 to 5 years and 6 or more years after fire the vegetation units had a higher probability of moderate and light grazing, respectively. Five simulations, each representing 85 years, were performed in SIMPPLLE following an initial 250 year run of the model. The initial 250 year run was designed to allow the modeling landscape to acclimate to the disturbance regimes, an important feature considering the starting ecosystem conditions of the landscape were based on generalized descriptions of the PLSR. In each of the simulations, climate (in other words, precipitation and temperature) patterns were varied within the range of climate patterns recorded for the Thunder

Basin area. Following the simulations the results were summarized by the historical states within each ecological site occurring within the planning area.

RESULTS AND DISCUSSION

Terrestrial plant community states for the Northern Rolling High Plains, Major Land Resource Area are fully described in Haufler and others (2008). These state descriptions include information about the species composition, structure, productivity, and the characteristic disturbances that occurred within each historical state. The average, minimum, and maximum number of ac historically occupied by each historical state in the Thunder Basin region can be found in table 1. A majority of the terrestrial ecosystems in this area, 80.30 percent, were in vegetation conditions described by the short interval fire regime (table 2) generally experiencing a fire at least once every 25 years. A smaller component of the terrestrial ecosystems (19.70 percent) were classified as having long fire return intervals (table 2). Of this, 9.75 percent of the grass/shrub ecosystems were sparsely vegetated, independent of grazing regimes, and would not support short fire return intervals (table 2).

Table 1—Results of the Historical Range of Variability (HRV) analysis for ecological sites within the Thunder Basin planning landscape in eastern Wyoming. The mean, minimum, and maximum number of acres historically occupied by a given state is presented for each ecological site.

Fire Return Interval (FRI)/ Grazing Regime		Ecological Site									
		HRV	Clayey	Shallow Clayey	Loamy	Shallow Loamy	Sands/ Sandy	Shallow Sandy	Shallow Hilly	Very Shallow	Saline Upland
Short FRI											
Light Grazing	Mean	40,470	44,690	90,235	20,566	20,285	10,327	9,435	— ^a	—	—
	Min	30,710	32,612	73,170	14,125	15,545	6,308	6,480			
	Max	47,969	53,490	102,646	25,538	25,024	13,914	11,716			
Moderate Grazing	Mean	44,161	65,569	77,307	17,628	21,043	10,575	8,087	—	—	—
	Min	40,113	59,184	70,326	15,312	17,252	7,421	7,025			
	Max	52,017	75,404	88,684	22,148	26,162	14,780	10,161			
Heavy Grazing	Mean	31,543	59,357	70,844	16,837	12,007	7,297	7,724	—	—	—
	Min	22,140	47,106	55,330	12,713	7,267	4,391	5,832			
	Max	43,327	76,612	91,269	22,205	18,705	11,008	10,187			
Long FRI											
Light Grazing	Mean	2,381	2,071	11,983	1,413	7,141	2,257	648	913	4,160	N/A ^b
	Min	952	690	6,981	452	3,665	155	207	518	3,353	
	Max	4,642	3,969	17,582	2,995	11,880	6,648	1,374	1,210	5,136	
Moderate Grazing	Mean	357	518	7,239	17	1,959	340	8	1,007	7,173	N/A
	Min	0	173	6,464	0	632	0	0	710	3,523	
	Max	1,309	1,553	10,601	57	4,108	2,041	26	1,444	12,309	
Heavy Grazing	Mean	119	345	1,034	45	758	124	21	682	31,113	N/A
	Min	0	0	259	0	126	0	0	406	25,552	
	Max	714	1,035	2,844	226	2,022	1,237	104	1,056	35,230	
Sparsely Vegetated		6,606	51,811	9,745	6,041	1,490	4,270	4,348	189	3,456	6499

^aEcological sites that did not have short fire return intervals because of sparse fuel.

^bInformation not available.

Historical vegetation conditions reflective of light, moderate, and heavy grazing regimes were found in near equal proportions in the Thunder Basin planning area with 31.36 percent, 30.85 percent, and 28.04 percent of plant communities classified as having compositions characteristic of light, moderate, and heavy grazing respectively (table 2). Of the terrestrial ecosystems classified as having a short fire return interval, 27.54 percent were characteristic of light grazing regimes, 28.74 percent were characteristic of moderate grazing regimes, and 24.02 percent were characteristic of heavy grazing regimes (table 2). For the terrestrial ecosystems classified as having a long fire return interval, 3.82 percent were characteristic of light grazing regimes, 2.11 percent were characteristic of moderate grazing regimes, 4.02 percent were characteristic of heavy grazing regimes, and 9.75 percent were sparsely vegetated regardless of grazing regimes (table 2).

The results of this historical range of variability simulation (tables 1 to 2) provide estimates of the historical ecosystem diversity within the Thunder Basin planning area. These results along with the model input information provide an estimate of reference conditions for historical states for each ecological site occurring in the Thunder Basin planning area. Various parameters within the SIMPPLLE model can be adjusted which would influence the model's results and thus the estimated amounts of plant community states. For example, increasing the number of lightning strikes will increase the number of fire starts. While the primary influence of fire was more related to its ability to spread across the landscape, changes in number of lightning strikes would result in some change in the percentage of the landscape in long or short-interval fire regimes. Similarly, changes to weightings for probabilities of grazing pressure would result in shifts in the percentages of the landscape in light, moderate, or heavy grazing regimes. The number of bison and other grazers that historically occurred in the landscape is not known. Therefore we estimated possible grazing effects to maintain some representation of all regimes. The specific amounts of each grazing regime that occurred historically may have differed from our estimates, but we have attempted to simulate a likely landscape configuration based on the best available information we complied.

Comparison of historical ecosystem conditions of this area with previously assessed existing ecosystem conditions (Ganguli and others 2008) reveal several interesting differences, most notably the widespread presence of exotic plant species especially *Bromus tectorum* and *Bromus arvensis*, the lack of native plant species that typically decrease with increasing grazing pressure, and the alteration of fire regimes. Through comparisons of historical ecosystem diversity with existing conditions we identified

specific plant communities that are underrepresented in the Thunder Basin planning area and in need of restoration to provide desired wildlife habitat and other altered ecosystem services. We have identified the information required to develop a coarse filter approach for conservation planning efforts and suggest that private and public land managers within the Thunder Basin planning area consider an integrated coarse and fine filter approach as they develop specific conservation objectives and a management plan for this area.

Table 2—Summary of the SIMPPLLE historical range of variability model simulation illustrating the percent of the Thunder Basin planning area that was classified, based on plant species composition, as being characteristic of disturbance (i.e., fire and grazing) induced historical states or states that are naturally sparsely vegetated.

Grazing Regime	Fire Return Interval	
	Short	Long
	% of the landscape	
Light	27.54	3.82
Moderate	28.74	2.11
Heavy	24.02	4.02
Sparsely Vegetated		9.75

Coarse filter and fine filters are terms that have been widely used to characterize different conservation strategies. Coarse filter strategies focus on providing an appropriate mix of ecosystems or ecological communities across a planning landscape, while fine filter strategies focus on providing for the needs of individual or multiple species within a landscape (Haufler 1999; Marcot and others 1994; Schwartz 1999; The Nature Conservancy 1982). The fundamental difference in the two conservation approaches is whether the primary basis of a strategy is focused on ecosystems or species. Coarse filter approaches assume that providing adequate ecosystem representation will maintain ecosystem integrity, habitat needs, and future persistence of all native species. However, simply providing specified amounts of each native or historically-occurring ecosystem in a landscape may not be sufficient because the size, juxtaposition, or distribution of the habitat patches provided by the representation conditions might not meet the needs of many species. Various conservation planning initiatives have moved towards using a combination of coarse and fine filter approaches to address their objectives so that both ecosystems and species are considered in the planning process.

IMPLICATIONS

Given the difficulty of identifying appropriate native ecosystem reference sites that are free of modern anthropogenic influences, spatially explicit landscape-scale modeling efforts can provide valuable information that can be used in conservation planning and management of grass and shrub ecosystems. A modeling tool like SIMPPLLE allows users to define the spatial scale (in other words, grain and extent) appropriate for a given modeling exercise and incorporate the best available information for an area. SIMPPLLE can also be used in simulations of current or future conditions allowing for investigations of plant community response to disturbance regime modification, climate change, and invasive plant species. In fact, SIMPPLLE has recently been used in a collaborative modeling effort to address landscape impacts to alternative management scenarios (Turner and others 2008). Tools like NRCS Ecological Site Descriptions and associated state and transition models often provide valuable information that can be used to characterize the SIMPPLLE model for simulations aimed at characterizing reference conditions. Furthermore, the integration of model-derived reference sites with information from ecological site descriptions can provide a useful resource for ecosystem restoration planning. To aid management and conservation planning efforts future development and modification of Ecological Site Descriptions should include as much information as possible about native or historical ecosystems that occurred as a function of historical disturbance regimes. Ecological site descriptions should also be further developed to provide better wildlife habitat information for historical and existing ecosystem states.

ACKNOWLEDGMENTS

Funding for this project was through grants provided to the Thunder Basin Grasslands Prairie Ecosystem Association by the U.S. Fish and Wildlife Service, Natural Resource Conservation Service, Powder River Coal Company, National Fish and Wildlife Foundation, Bureau of Land Management, U.S. Forest Service, Leopold Stewardship Fund of the Sand County Foundation, Sonoran Institute, Wyoming Governor's Federal Policy Fund, Rio Tinto Energy America (previously Kennecott Energy Company), Converse and Weston Counties of Wyoming, and a grant to EMRI from the Bradley Fund for the Environment. We appreciate the support and assistance given by Scott Yeats of the Ecosystem Management Research Institute, private landowners within the Thunder Basin planning area, and USFS personnel, namely Rebecca Schreiner and Kirk Moeller. A special thank you to the Science Team that included Everett Bainter, Clarke McClung, Kyle Schmitt, Stan Boltz, Kurt Hansen, Jim O'Rourke, and Sam Fuhlendorf, who provided input to the assumptions and indicator species used in the development of the historical range of variability for the Thunder Basin region.

REFERENCES

- Anderson, R.C. 1990. The historic role of fire in the North American grassland. p. 8-18. *In*: Collins, S.L., Wallace, L.L., eds. *Fire in North American Tallgrass Prairies*. University of Oklahoma Press, Norman.
- Aplet, G.H., Keeton, W.S. 1999. Application of historical range of variability concepts to biodiversity conservation. p. 71-86. *In*: Baydack, R.K., Campa, H., Haufler, J.B. eds. *Practical approaches to the conservation of biological diversity*. Island Press, Washington, D.C.
- Bamforth, D.B. 1987. Historical documents and bison ecology on the Great Plains. *Plains Anthropologist* 32:1-16.
- Barrett, T.M. 2001. Models of vegetation change for landscape planning: a comparison of FETM, LANDSUM, SIMPPLLE, and VDDT. Gen. Tech. Rep. RMRS-GTR-76-WWW. Ogden, UT: USDA, Forest Service, Rocky Mountain Research Station. 14p.
- Biondini, M.E., Steuter, A.A., Hamilton, R.G. 1999. Bison use of fire-managed remnant prairies. *Journal of Range Management* 52:454-461.
- Bragg, T.B., Hulbert, L.C. 1976. Woody plant invasion of unburned Kansas bluestem prairie. *Journal of Range Management* 29:19-24.
- Chew, J.D., Stalling, C., Moeller, K. 2004. Integrating knowledge for simulating vegetation change at landscape scales. *Western Journal of Applied Forestry* 19:102-108.
- Coppedge, B.R., Shaw, J.H. 1998. Bison grazing patterns on seasonally burned tallgrass prairie. *Journal of Range Management* 51:258-264.
- Daubenmire, R. 1968. *Plant communities: a textbook of plant synecology*. Harper and Row, New York, NY.
- Engle, D.M., Bidwell, T.G. 2001. Viewpoint: The response of central North American prairies to seasonal fire. *Journal of Range Management* 54:2-10.
- Fisher, R.F., Jenkins, M.J., Fisher, W.F. 1987. Fire and the prairie-forest mosaic of Devils Tower National Monument. *American Midland Naturalist* 117:250-257.
- Forbis, T.A., Provencher, L., Turner, L., Medlyn, G., Thompson, J., Jones, G. 2007. A method for landscape-scale vegetation assessments: applications to Great Basin rangeland ecosystems. *Rangeland Ecology and Management* 60:209-217.
- Frank, D.A., McNaughton, S.J., Tracy, B.F. 1998. The ecology of the earth's grazing ecosystems. *BioScience* 48:513-521.
- Fuhlendorf, S.D., Engle, D.M. 2001. Restoring heterogeneity on rangelands: ecosystem management based on evolutionary grazing patterns. *BioScience* 51:625-632.
- Fuhlendorf, S.D., Engle, D.M. 2004. Application of the fire-grazing interaction to restore a shifting mosaic on tall grass prairie. *Journal of Applied Ecology* 41:604-614.
- Ganguli, A.C., Haufler, J.B., Mehl, C.A., Yeats, S.D. 2008. Ecological assessment of sagebrush grasslands in eastern Wyoming. Abstracts, the Fifteenth Wildland Shrub Symposium, Wildlands and Wildlife Habitat, 1998 June 17-19, Bozeman, MT.
- Hann, W.J. 2003. Reference conditions for Northern Plains Grassland (with shrubs) and Northern Plains Grasslands (without shrubs). *In*: Interagency and The Nature Conservancy fire regime condition class website (<http://www.frcc.gov>). The Nature Conservancy and Systems for Environmental Management.
- Haufler, J.B., Mehl, C.A., Ganguli, A.C., Yeats, S.D. 2008. Thunder Basin Grasslands: Ecological Assessment of Terrestrial Systems. Ecosystem Management Research Institute Technical Report. Available on-line at: www.emri.org.
- Haufler, J.B. 1999. Strategies for conserving terrestrial biological diversity. p. 17-30. *In*: Baydack, R.K., Campa, H., Haufler, J.B. editors. *Practical approaches to the conservation of biological diversity*. Island Press, San Diego, CA.
- Haufler, J.B. 2000. Ecosystem management: from rhetoric to reality. *Transactions of the North American Wildlife and Natural Resource Conference* 65:11-33.
- Haufler, J.B., Baydack, R.K., Campa, H., Kernohan, B.J., O'Neil, L.J., Waits, L., Miller, C. 2002. Performance measures for ecosystem management and ecological sustainability. *The Wildlife Society Technical Review* 02-1. Bethesda, Maryland.
- Haufler, J.B., Mehl, C.A., Roloff, G.J. 1996. Using a coarse-filter approach with species assessment for ecosystem management. *Wildlife Society Bulletin* 24:200-208.

- Higgins, K.F. 1984. Lightning fires in North Dakota Grasslands and in pine-savanna lands of South Dakota and Montana. *Journal of Range Management* 37:100-103.
- Hoch, G.A., Briggs, J.M., Johnson, L.C. 2002. Assessing the rate, mechanisms, and consequences of the conversion of tallgrass prairie to *Juniperus virginiana* forest. *Ecosystems* 5:578-586.
- Knapp, A.K., Blair, J.M., Briggs, J.M., Collins, S.L., Hartnett, D.C., Johnson, L.C., Towne, E.G. 1999. Keystone role of bison in North American tallgrass prairie. *BioScience* 49:39-50.
- Knight, D.H. 1994. *Mountains and Plains: The Ecology of Wyoming Landscapes*. Yale University Press, New Haven, CT.
- Komarek, E.V. 1964. The natural history of lightning. Proceedings of the 3rd annual Tall Timbers fire ecology conference. Tall Timbers Research Station, Tallahassee, FL 3:139-148.
- Kotliar, N.B., Baker, B.W., Whicker, A.D., Plumb, G.E. 1999. A critical review of assumptions about the prairie dog as a keystone species. *Environmental Management* 24:177-192.
- Marcot, B., Wisdom, M.J., Li, H.W., Castillo, G.C. 1994. Managing for featured, threatened, endangered, and sensitive species and unique habitats for ecosystem sustainability. USDA. General Technical Report PNW-GTR-329.
- McNaughton, S.J. 1985. Ecology of a grazing ecosystem. *Ecological Monographs* 55:259-294.
- Miller, B., Ceballos, G., Reading, R. 1994. The prairie dog and biotic diversity. *Conservation Biology* 8:677-681.
- Morgan, P., Aplet, G.H., Haufler, J.B., Humphries, H.C., Moore, M.M., Wilson, W.D. 1994. Historical range of variability: a useful tool for evaluating ecosystem change. *Journal of Sustainable Forestry* 2:87-112.
- Nilsson, C., Jansson, R., Malmqvist, B., Naiman, R.J. 2007. Restoring riverine landscapes: the challenge of identifying priorities, reference states, and techniques. *Ecology and Society* 12:16. <http://www.ecologyandsociety.org/vol12/iss1/art16/>.
- Perryman, B.L. 1996. The vegetation mosaic of the Rochelle Hills area of the Thunder Basin National Grasslands. In: *Fire history of the Rochelle Hills area of the Thunder Basin National Grasslands*. PhD Dissertation, University of Wyoming, Laramie.
- Perryman, B.L., Laycock, W.A. 2000. Fire history of the Rochelle Hills Thunder Basin National Grasslands. *Journal of Range Management* 53:660-665.
- Poiani, K.A., Richter, B.D., Anderson, M.G., Richter, H.E. 2000. Biodiversity conservation at multiple scales: Functional sites, landscapes, and networks. *Bioscience* 50:133-146.
- Schwartz, M.W. 1999. Choosing the appropriate scale of reserves for conservation. *Annual Review of Ecological Systems* 30:83-108.
- Shinneman, D.J., Baker, W.L., Lyon, P. 2008. Ecological restoration needs derived from reference conditions for a semi-arid landscape in Western Colorado, USA. *Journal of Arid Environments* 72:207-227.
- Sieg, C.H., Flather, C.H., McCanny, S. 1999. Recent biodiversity patterns in the Great Plains: implications for restoration and management. *Great Plains Research* 9:277-313.
- Soper, J.D. 1941. History, range and home life of the northern bison. *Ecological Monographs* 11:347-412.
- Stoddard, J.L., Larson, D.P., Hawkins, C.P., Johnson, R.K., Norris, R.H. 2006. Setting expectations for the ecological condition of streams: the concept of reference condition. *Ecological Applications* 16:1267-1276.
- Templeton, A.R., Robertson, R.J., Brisson, J., Strasburg, J. 2001. Disrupting evolutionary processes: the effect of habitat fragmentation on collard lizards in the Missouri Ozarks. *Proceedings of the National Academy of Sciences* 98:5426-5432.
- The Nature Conservancy. 1982. *Natural heritage program operations manual*. The Nature Conservancy.
- Turner, C.E., Romme, W.H., Chew, J., Miller M.E., Leavesley, G., Floyd-Hanna, L., San Miguel, G., Cobb, N.S., Zirbes, R., Viger, R., Ironside, K.E. 2008. The FRAME Project—A Collaborative Modeling Approach to Natural Resource Management at Mesa Verde National Park, Colorado. In: Van Riper, C., III, and M. K. Sogge, eds. *The Colorado Plateau III: Integrating Research and Resources Management for Effective Conservation*, University of Arizona Press, Tucson. p 23-41
- United States Department of Agriculture, Natural Resources Conservation Service. 2003. *National range and pasture hand book*. Washington, D.C.
- Whicker, A.D., Detling, J.K. 1988. Ecological consequences of prairie dog disturbances. *BioScience* 38:778-785.
- Williams, G.W. 2005. References on the American Indian use of fire in ecosystems. In *Report prepared for the USDA Forest Service*, Washington, D.C. 129p.
- Wuerthner, G. 1997. Viewpoint: The black-tailed prairie dog-headed for extinction? *Journal of Range Management* 50:459-466.

Short-Term Response of Shrubs, Graminoids, and Forbs to Mechanical Treatment in a Sagebrush Ecosystem in Colorado

Adam C. Payton¹, Sandra J. Hayes¹, Sandra M. Borthwick²,
and Russell D. Japuntich¹

ABSTRACT

Declines in Gunnison sage grouse populations are thought to be related to habitat loss, fragmentation, and human induced habitat changes. In an attempt to improve the quality of early brood rearing habitat the Bureau of Land Management, Gunnison Field Office in Gunnison Colorado implemented a series of mechanical treatments designed to reduce sagebrush canopy cover and increase cover of graminoids and forbs. Brush mowing and Dixie harrow were utilized in 2005 to treat 30 percent of six 14 ac sites. In 2006 and 2007 shrub canopy cover, graminoid cover, forb cover, heights, and species richness were assessed to determine the vegetative response to each treatment. Sagebrush canopy cover was reduced to approximately 15 percent by both treatments. Mowing appeared to have no effect on forb or non-sagebrush shrub canopy cover, however, graminoid cover increased slightly post-treatment. Sites treated with the Dixie harrow had increased non-sagebrush shrub canopy cover, graminoid cover, and forb cover in post treatment years as compared to pretreatment. Heights for graminoids and forbs did not differ pre and post treatment ($p > 0.05$) nor did species richness ($p > 0.05$) for either treatment.

INTRODUCTION

The recently described Gunnison sage grouse (*Centrocercus minimus*) inhabits southwestern Colorado and eastern Utah. This species is a sagebrush (*Artemisia* sp.) obligate whose population has been declining due to habitat loss, fragmentation, and human induced habitat changes (Oyler-McCance and others 2001; Young and others 2000). The historic range of the Gunnison sage grouse is thought to include Grand and San Juan counties in Utah, fifteen southwestern counties in Colorado, and northern counties in New Mexico (Braun 1995; Young and others 2000). Currently the distribution has been reduced to six counties in Colorado and one in Utah comprising eight known populations with a total of fewer than 5000 breeding individuals (Young and others 2000). The largest of these is the Gunnison Basin population located in Gunnison County Colorado, containing approximately 3700 breeding individuals (Spicer and Diamond 2007).

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Adam Payton, Sandra J. Hayes, Russell D. Japuntich; Bureau of Land Management, Gunnison Field Office, Gunnison Co

²Sandra M. Borthwick; Bureau of Land Management, Henry Mountain Field Station, Hanksville UT

Habitat use by sage grouse (*Centrocercus* sp.) varies temporally and spatially. Sagebrush cover, herbaceous cover, and food availability, which is dictated by sagebrush and herbaceous cover, are the characteristics that drive habitat selection by nesting hens and hens with broods. Nesting habitats are typically distinguished by having 17 to 22 percent canopy cover of sagebrush which exhibits a greater percent cover than sites randomly selected which contain 7 to 15 percent sagebrush cover (Connelly 1991; Sveum 1998; Klebenow 1969; Gregg 1991). Increased levels of sagebrush cover appear to be preferred during nesting where concealment is a desired habitat characteristic. However following hatching habitat use shifts to areas with higher food availability.

The primary dietary component for young sage grouse consists of forbs and insects (Klebenow 1969; Drut 1994b, Peterson 1970). The availability of their primary food becomes the driving force for habitat selection for both early and late brood rearing. Early brood rearing, zero to six weeks, represents a habitat transition from sagebrush dominated sites used for nesting to riparian, meadow, or agricultural areas where food is more abundant. Brood rearing sites are typically characterized by a decrease in shrub canopy cover to 10 to 15 percent and an increase in forb cover, 10 to 27 percent (Connelly and others 2000; Drut 1994a, Klebenow 1969; Gunnison Sage-grouse Rangewide Steering Committee 2005).

Severe declines in Gunnison sage grouse populations resulting from loss of habitat and decreased habitat quality prompted the Bureau of Land Management Gunnison Field Office to implement habitat improvement projects. In an effort to improve early brood rearing habitat areas of big sagebrush (*Artemesia tridentata*) were treated using either a tractor pulled brush mower or Dixie harrow. Both treatments were designed to reduce sagebrush canopy cover releasing forbs and grasses from competition with sagebrush. Little research currently exists evaluating the response of grasses, forbs, and shrubs of a sagebrush system to mechanical treatment (Dahlgren 2006). The effects of sagebrush mowing and harrowing have rarely been documented despite their widespread use (Dahlgren 2006). As the prevalence of sagebrush treatments increases on BLM land, we were interested in the short-term vegetative response associated with mowing and harrowing as it applies to improving the condition of sage grouse habitat.

METHODS

Study Site

The treatment sites were located on lands managed by the Bureau of Land Management's Gunnison Field Office located in Gunnison, Colorado (38° 32' 45" N, 106° 55' 54" W). The sites were in the Sage Hen pasture of South Parlin Flats located 22.5km (14 miles) south east of Gunnison (figure 1). The sites are generally characterized as high elevation, 2500m (8200 ft), low precipitation, 13.7cm during the course of the study, with shrub communities comprised of a mosaic of mountain big sagebrush (*Artemisia tridentata* spp. *vaseyana*) dominated sites and Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) dominated sites. The pasture is typically grazed by cattle but grazing was suspended during the time of the study. Within the Sage Hen pasture there is an inactive lek last utilized in 1995. An active lek is located approximately 200m (1/8 mile) to the northeast of the pasture with portions of the treatment area utilized by sage grouse for both summer and winter habitat.

Vegetation Treatments

The Dixie harrow utilized consisted of a 4.2m (14 foot) wide draw bar with ten 3m long heavy steel tubes attached to it. Each tube has many alternating 30cm (12 inch) steel fins attached. As the harrow is dragged behind a tractor the fins catch on plants and either uproot entire sagebrush plants or breaks portions off leaving plants with a reduced or altered canopy. A broadcast seeder was attached to the front of the tractor dispensing seed in the path of the harrow, which disrupts the soil surface adequately enough to assist in seedbed preparation. A grass/forb seed mix comprised of Sandberg bluegrass (*Poa secunda*), bottlebrush squirreltail (*Elymus elymoides*), western wheatgrass (*Pascopyrum smithii*), Indian ricegrass (*Achnatherum hymenoides*) alfalfa (*Medicago* sp.), small burnet (*Sanguisorba minor*), blue flax (*Linum perenne*), and sulfur buckwheat (*Eriogonum umbellatum*) was broadcast seeded at a rate of 4.4 kg (10 lbs) per ac. Three sites were selected for treatment by Dixie harrow, each approximately 14 ac. Thirty percent of each site was treated using a single pass from the harrow. Serpentine paths were created to reduce the potential threat of predation from ground predators and mimic a more natural pattern of disturbance.

The brush mowing treatment consisted of an approximately 4.6 m (15 ft) wide tractor-pulled mower set to a height of 20.3 cm (8 inches). The previously described grass/forb seed mix was dispensed at a rate of 4.4 kg (10 lbs) per ac from a seeder mounted in front of the tractor. Compared to harrowing, mowing resulted in less soil surface disturbance and thus was less effective at preparing a seedbed. Three sites were selected for treatment by brush mowing, each approximately 14 ac. Thirty percent of each site was treated in a serpentine pattern using a single pass from the mower.

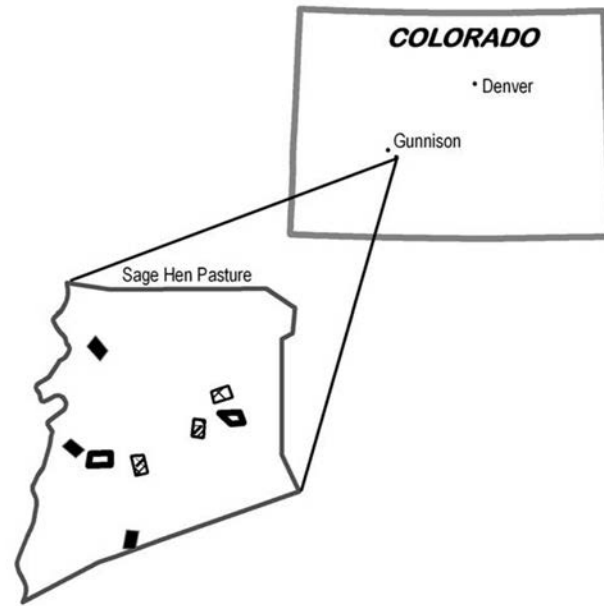


Figure 1—Location of treatment sites in Sage Hen Pasture near Gunnison Colorado. Solid fill sites are mowing, cross hatched sites are Dixie harrow, and open fill sites are controls.

Vegetation Analysis

One 30 m transect was established per treatment site by permanently marking the beginning and end points. Each transect was read pretreatment in 2005. Mechanical treatment was conducted in the fall of 2005 with each transect being marked and deliberately treated for its entire length. Two control sites were established in similar vegetation types as the treatments. All transects were read post treatment in August 2006 and 2007.

Estimates of cover for sagebrush (*Artemisia tridentata*) and non-sagebrush shrubs (*Chrysothamnus* sp., *Tetradymia canescens*, *Opuntia polyacantha*, and *Symphoricarpos rotundifolius*) were determined by measuring the horizontal linear length of live and dead portions of shrub plants using line-intercept methodology. Measurements were made to the nearest 3 cm (1/10 ft) excluding spaces of 6cm (2/10 ft) and smaller between branches or foliage with live plant material taking precedence over dead when overlapping occurred.

Canopy cover for graminoids and forbs was determined by estimating the cover by species and lifeform of a 20 x 50 cm (7.8 x 19.6 inch) microplot placed every three m along the transect. Estimates of canopy cover were placed into one of thirteen cover classes (0 to 1 percent, 1 to 5 percent, 5 to 10 percent, 10 to 20 percent, 20 to 30 percent, 30 to 40 percent, 40 to 50 percent, 50 to 60 percent, 60 to 70 percent, 70 to 80 percent, 80 to 90 percent, 90 to 99 percent, 99 to 100 percent), which were averaged to determine percent cover per transect. In addition, estimates of ground cover

(live and standing dead vegetation, litter, gravel, cobble, stones, and bedrock) and bare soil were also recorded. Average height, measured as the height where the bulk of the live and residual foliage occurs, maximum height, measured as the height of the tallest naturally occurring leaf or inflorescence (in the case of forbs), and density were recorded for each species encountered within the microplot. This paper will address shrub, graminoid, and forb canopy cover as well as height comparisons and species richness.

Statistical Analysis

Percent change, calculated as $A-B/B$, was used to determine if a change existed between pre and post treatment shrub, graminoid, and forb canopy cover. Since only one cover value was assigned per transect it generates a small sample size ($n = 3$) which precluded a statistical analysis. However height measurements generated a larger sample size since each measurement is independent allowing for a single factor analysis of variance (ANOVA) to be performed to determine if differences in height existed within species between pre and post treatment. Species richness was determined per transect and compared using a Wilcoxon signed-rank test to determine if a difference existed between pre and post treatments.

RESULTS

Sagebrush Canopy Cover

Mowing reduced the amount of sagebrush canopy cover along the transects from 21.9 percent pretreatment in 2005 to 16.4 percent and 14.8 percent for 2006 and 2007 respectively (figure 2). This represents a canopy reduction of 25.2 percent between 2005 and 2006 and 32.2 percent between 2005 and 2007. Dixie harrow also reduced the sagebrush canopy along the transects from 37.4 percent in 2005 to 13.7 percent in 2006 with an increase to 16.9 percent in 2007. This represents a decrease in canopy cover of 63.3 percent between 2005 and 2006 and a decrease of 54.9 percent between 2005 and 2007. Controls showed a reduction in sage canopy cover of 2.9 percent in both 2006 and 2007 (table 1).

Non-Sagebrush Shrub Canopy Cover

Non-sagebrush shrub canopy cover along the transects was increased by mowing from 1.4 percent pretreatment in 2005 to 2.2 percent and 2.8 percent in 2006 and 2007, respectively. This represents a 61.0 percent increase between 2005 and 2006 and a 104.9 percent increase between 2005 and 2007 (figure 2). Dixie harrow increased non-sagebrush canopy cover along the transects from 7.3 percent in 2005 to 12.5 and 13.9 percent in 2006 and 2007. This represents a 71.0 percent increase between 2005 and 2006 and 89.2 percent increase between 2005 and 2007. The control's non-sagebrush canopy cover increased from 6.9 percent in 2005 to 8.0 and 8.3 percent in 2006 and 2007.

This represents a 16.1 percent increase in 2006 and a 20.4 percent increase between 2005 and 2007 (table 1).

Graminoid Canopy Cover

Graminoid canopy cover along the transects in mowed treatments decreased from 33.5 percent in 2005 pretreatment measurements to 22.1 percent post treatment in 2006 a decrease in cover of 34.2 percent. It then increased to 39.7 percent in 2007 representing an increase of 18.3 percent when comparing 2005 to 2007. Graminoid canopy cover along the transects in Dixie harrow treatments also declined post-treatment from 11.7 percent in 2005 to 6.7 percent in 2006 a decrease of 43.3 percent. In 2007 canopy cover along the transects increased to 26 percent representing a 121.9 percent increase between 2005 and 2007 (figure 3). Controls also experienced a decline in graminoid canopy cover between 2005 and 2006. Cover decreased 42.0 percent from 27.4 to 15.9 percent with an increase in 2007 to 23.8 percent but canopy cover remained 13.1 percent lower than pretreatment measurements (table 1).

Forb Canopy Cover

Forb canopy cover, consisting of both annual and perennial forbs, decreased along the transects in mowed treatments from pretreatment levels of 3.9 percent to 3.0 percent in 2006 a decrease of 21.9 percent cover. Forb cover increased in 2007 to 3.3 percent representing an overall decrease of 16.3 percent between 2005 and 2007. Forb canopy cover in Dixie harrow treatments increased from 0.9 percent in 2005 to 7.0 percent in 2006 this represents a 679.6 percent increase. In 2007 forb cover decreased to 4.8 percent along the transects, representing an overall forb canopy cover increase of 427.8 percent between 2005 and 2007 (figure 3). Forb canopy cover of controls decreased from 11.8 percent in 2005 to 9.4 percent, a decrease of 20.3 percent. In 2007 canopy cover increased slightly to 9.7 percent but cover remained 17.8 percent lower than in 2005 (table 1).

Graminoid and Forb Height

The average bulk height of both graminoids and forbs, including annuals and perennials, for each treatment (mowing, Dixie harrow, and control) showed no significant difference between pre and post treatment heights ($P > 0.05$).

Species Diversity

Shrub, graminoid, and forb species richness did not significantly change during the two years post-treatment in any transect regardless of the treatment ($P > 0.10$). In 2006 seeded forb species were detected in five transects consisting of alfalfa found along one transect and small burnet found along four transects. These individuals were not subsequently detected along the transects in 2007. No detection of any seedling seeded grass species occurred along the transects in 2006 or 2007.

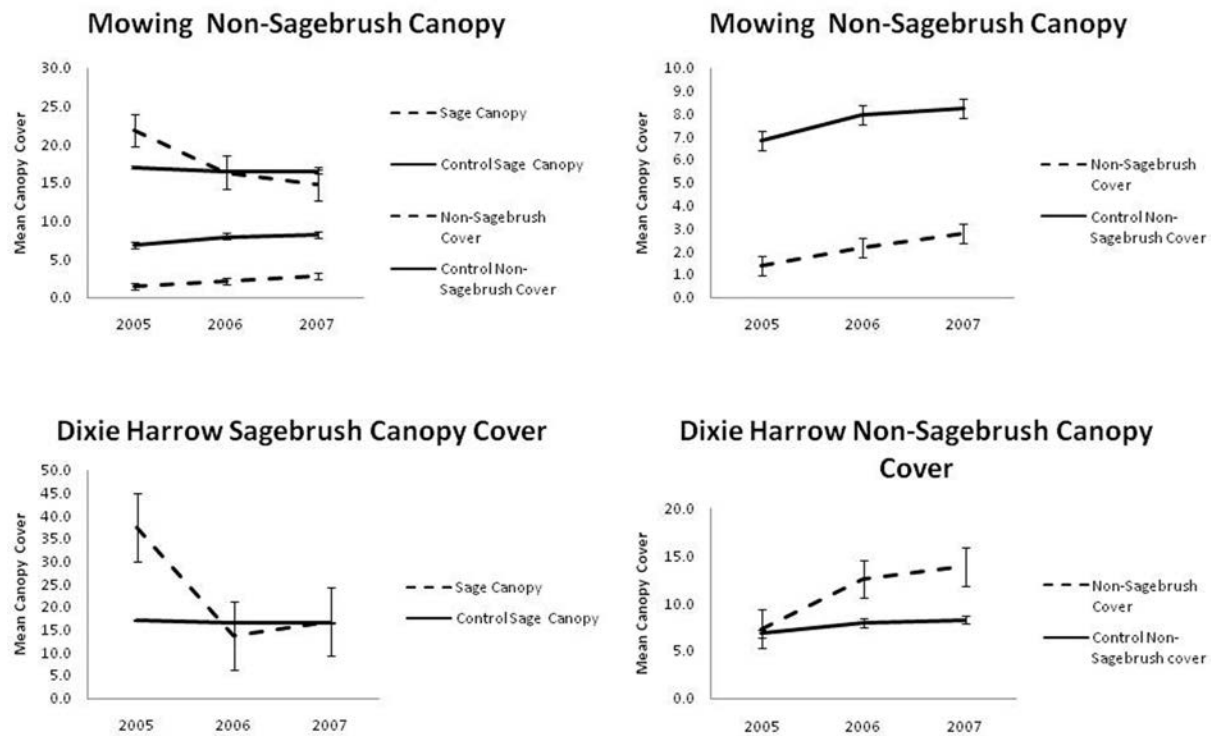


Figure 2—Mean canopy cover of live sagebrush and live non-sagebrush shrubs by treatment type. Error bars represent standard error.

DISCUSSION

Sagebrush canopy cover was reduced by both mowing and Dixie harrow, potentially allowing for competitive release of graminoids and forbs to improve early brood rearing habitat for the Gunnison sagegrouse. Controls maintained a consistent canopy cover over the course of the study indicating that the decrease seen from mowing and harrowing can be attributed to the treatment. In 2007 sagebrush in mowed transects experienced a continued decline in canopy cover while sagebrush in harrowed transects increased, by 97.5 cm (3.2 ft). The decrease seen in the mowing represents continued shrub dieback between 2006 and 2007. However sagebrush in harrowed sites increased in cover between 2006 and 2007 indicating that the plants responded to treatment. This increase may be seen as an unusually high change in a single growing season but given that transect cover values are compiled from numerous measurements of individual plants a relatively small increase that is seen uniformly across many individuals can quickly increase the overall cover of a site.

Non-sagebrush shrub canopy cover data, while containing measurements from four species, is predominantly comprised of rabbitbrush (*Chrysothamnus* sp.). Mowing created an increase of non-sagebrush shrub canopy cover of 104.9 percent between 2005 and 2007. Controls showed an increase of 20.4 percent between 2005 and 2007. When strictly comparing the mowing treatment percent difference

with the control it would appear that mowing created a larger increase of non-sagebrush shrub cover. However, due to the initially low pretreatment canopy cover of 1.4 percent for mowed sites the increase to 2.8 percent cover will return a large percent change but this represents an artificially inflated representation of treatment effect. Comparison of the slopes of the lines, which indicates the degree of change from 2005 to 2007, offers a more accurate assessment of treatment effect. The slopes for mowed sites and controls are almost identical; $y=0.7x$ for the controls and $y=0.716x$ for mowing. This slope comparison indicates that there was almost no difference in the degree of change between mowed areas and controls for non-sagebrush canopy cover (figure 3). In Dixie harrow treatments non-sagebrush shrub canopy cover increased 89.2 percent between 2005 and 2007. Controls increased 20.4 percent between 2005 and 2007. Since the treatment and the control mean canopy cover were very similar in 2005, 7.3 and 6.8 percent respectively, the change seen in the harrowed sites can be interpreted as an effect of the treatment.

Graminoid canopy cover decreased between 2005 and 2006 for mowing (-34.2 percent), Dixie harrow (-43.3 percent), and controls (-42.0 percent). This was then followed by an increase in cover for all three treatments between 2006 and 2007. This initial decline in 2006 coincides with an abnormally low amount of precipitation in July, a critical growing period, indicating that this universal decrease is the result of climatic variation. In 2007 graminoid cover for mowings had increased to 18.3 percent above 2005 values.

While this represents an overall increase there is sufficient overlap in the variation within the samples for 2005 and 2007 to view this increase with caution. Changes in graminoid canopy cover in response to mowing remains unclear and further observation is needed to determine if there is a treatment effect. Dixie harrow graminoid cover increased in 2007 by 121.9 percent over 2005 values. The mean cover values and the variation that exists within these values are sufficiently different that it indicates the increase in graminoid cover is due to treatment by Dixie harrow (figure 3). The graminoid cover in the controls also experienced an increase in 2007 but it was still -13.1 percent below 2005 canopy values. The response of the controls further validates that the initial decline in 2006 can be attributed to environmental conditions. It also suggests that the increase seen in harrowed sites is due to the treatment.

Annual and perennial forb canopy cover decreased for controls (-20.3 percent) and mowing (-21.9 percent) between 2005 and 2006 then remained essentially the same in 2007 (figure 3). The similarity in response between mowing and controls suggests that there was no effect of mowing on forb cover within the first two years post-treatment. Dixie harrow forb cover increased between 2005 and 2006 then declined in 2007. The dramatic increase in 2006 is partially due to a large number of annual forbs. The decline in 2007 incorporates a weaker production of annual forbs and is more representative of the treatment effect on perennial forb production. Again percent difference within the harrowed treatments must be interpreted with some caution since the beginning value is small, 0.9, and any

slight increase in cover will result in a dramatic change in percent difference. However it appears that within harrowed sites initially there is a trend for increased forb cover.

Seeding success for this study was marginal. No detection of seeded grass seedlings and only five individuals of forbs after two years suggests that seed germination and seedling survival is very low in this system, especially among non-local genotypes. Based on the low success rate at these sites, seeding may not provide any additional benefits in this ecosystem. This further reinforces the idea that if restoration of a site is desired it may be most effective to manage the area for the plants that already exist rather than trying to augment or change them through seeding.

These data represent the changes observed within a dynamic and complex sagebrush ecosystem two years post treatment. Two years is a short time-frame to assess the response of a plant community to disturbance. This should be kept in mind with interpreting these results. Continued study is required to fully understand how this particular sagebrush system responds to these disturbances.

ACKNOWLEDGEMENTS

There were numerous people involved in the methodology development and data collection of this study. We would like to thank the following for their contribution to the success of this study: Chris Cadwell, Suzy Parker, Joe Pecor, Vinnie Rossignol, and Scott Sherwood.

Table 1—Mean canopy cover of sagebrush, non-sagebrush, graminoids, and forbs by treatment by year. Percent difference represents difference between 2005, pretreatment, and 2006 and 2007, post-treatment.

Treatment	Sagebrush Cover	Percent Change	Non-Sagebrush Cover	Percent Change	Graminoid Cover	Percent Change	Forb Cover	Percent Change
Mowing 2005	21.9		1.4		33.5		3.9	
Mowing 2006	16.4	-25.2	2.2	61.0	22.1	-34.2	3.0	-21.9
Mowing 2007	14.8	-32.2	2.8	104.9	39.7	18.3	3.3	-16.3
Harrow 2005	37.4		7.3		11.7		0.9	
Harrow 2006	13.7	-63.3	12.5	71.0	6.7	-43.3	7.0	679.6
Harrow 2007	16.9	-54.9	13.9	89.2	26.0	121.9	4.8	427.8
Control 2005	17.0		6.9		27.4		11.8	
Control 2006	16.5	-2.9	8.0	16.1	15.9	-42.0	9.4	-20.3
Control 2007	16.5	-2.9	8.3	20.4	23.8	-13.1	9.7	-17.8

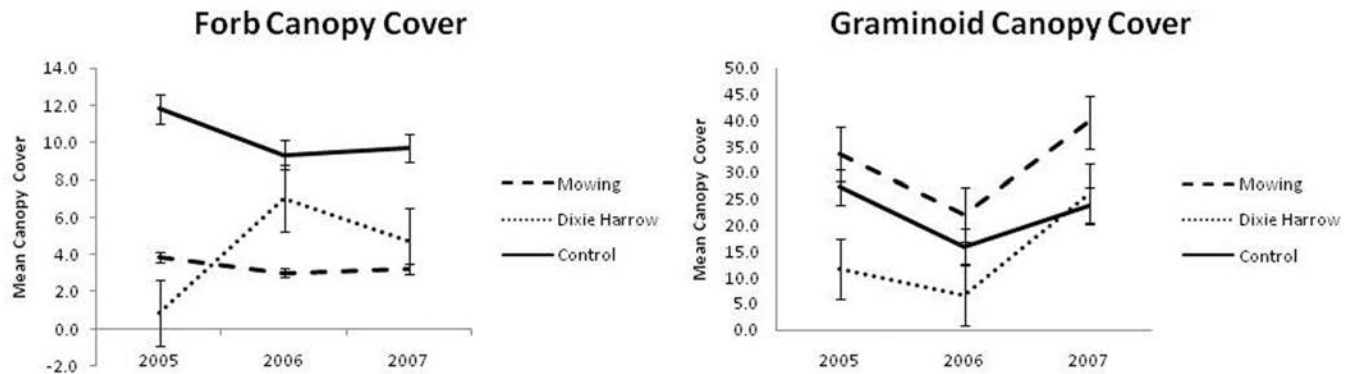


Figure 3—Mean canopy cover of graminoids and annual and perennial forbs by treatment. Error bars represent standard error.

REFERENCES

- Braun, C.E. 1995. Distribution and status of sage grouse in Colorado. *The Prairie Naturalist*. 27(1):2-9.
- Commons, M.L., Daybreak, R.K., Braun, C.E. 1997. Gunnison sage grouse *Centrocercus minimus* use of fragmented habitat in southwestern Colorado. *Wildlife Biology*. 3(4):283.
- Connelly, J.W., Schroeder, M.A., Sands, A.R., Braun, C.E. 2000. Guidelines to manage sage grouse populations and their habitats. *Wildlife Society Bulletin*. 28(4):967-985.
- Connelly, J.W., Wakken, W.L., Apa, A.D., Reese, K.P. 1991. Sage grouse use of nest sites in southeastern Idaho. *Journal of Wildlife Management*. 55(3):521-524.
- Dahlgren, D.K., Chi, R., Messmer, T.A. 2006. Greater sage-grouse response to sagebrush management in Utah. *Wildlife Society Bulletin*. 34(4): 975-985.
- Drut, M.S., Crawford, J.A., Gregg, M.A. 1994a. Brood habitat use by sage grouse in Oregon. *Great Basin Naturalist*. 54(2): 170-176.
- Drut, M.S., Pyle W.H., Crawford J.A. 1994b. Diets and food selection of sage grouse chicks in Oregon. *Journal of Range Management*. 47:90-93.
- Gregg, M.A. 1991. Use and selection of nesting habitat by sage grouse in Oregon. Thesis, Oregon State University, Corvallis, USA.
- Gunnison Sage-grouse Rangewide Steering Committee. 2005. Gunnison sage-grouse rangewide conservation plan. Colorado Division of Wildlife, Denver, Colorado.
- Hendrick, D.W., Hyder, D.N., Sneva, F.A., Poulton, C.E. 1966. Ecological response of sagebrush-grass range in central Oregon to mechanical and chemical removal of *Artemisia*. *Ecology*. 47(3):432-439.
- Klebenow, D.A. 1969. Sage grouse nesting and brood habitat in Idaho. *Journal of Wildlife Management*. 33(3):649-662.
- Oyler-McCance, S.J., Burnham, K.P., Braun, C.E. 2001. Influence of changes in sagebrush on Gunnison sage grouse in southwestern Colorado. *The Southwestern Naturalist*. 46(3):323-331.
- Peterson, J.G. 1970. Food habits and summer distribution of juvenile sage grouse in central Montana. *Journal of Wildlife Management*. 34(1): 147-155.
- Spicer, L., Diamond, B. 2007. Gunnison basin Gunnison sage-grouse lek count summary and population estimate. Colorado Division of Wildlife report.
- Sveum, C.M., Edge, D.W., Crawford, J.A., 1998. Nesting habitat selection by sage grouse in southcentral Washington. *Journal of Range Management*. 51(3):265-269.
- Wirth, T.A., Pyke, D.A. 2003. Restoring forbs for sage grouse habitat: fire, microsites, and establishment methods. *Restoration Ecology*. 11(3): 370-377.
- Young, J.R., Braun, C.E., Oyler-McCance, S.J., Hupp, J.W., Quinn, T.W. 2000. A new species of sage-grouse (Phasianidae: *Centrocercus*) from southwestern Colorado. *Wilson Bulletin*. 112(4): 445-453.

Displacement of Native Riparian Shrubs by Woody Exotics: Effects on Arthropod and Pollinator Community Composition

Rosemary L. Pendleton¹, Burton K. Pendleton¹, and Deborah Finch¹

ABSTRACT

Throughout the southwestern U.S., riparian gallery forests of cottonwood and willow are being invaded by woody exotics, primarily Russian olive and salt cedar. We wondered what effect this might have on native pollinator populations. Pollinators are indispensable contributors to biodiversity, ecosystem health, and human food production. Recent declines in pollinator abundance and health, such as catastrophic declines in honey bee populations due to Colony Collapse Disorder, has renewed interest in native pollinators and the ecosystem services they provide. Insects were collected from willow, Russian olive and salt cedar throughout April and May of 1997 and 1998 using sweep nets. For each collection day, nets were swept over the target shrubs for a specified number for passes to ensure equal collection effort. Insects were counted and identified to family. Total numbers were adjusted by number of sweep-days. Total insect abundance was greatest for willows (33.5 insects per sweep-day), followed by Russian olive (18.0) and salt cedar (6.8). Willows also had the greatest number of insect orders and families represented. Of the four primary insect pollinator orders, willow had the greatest numbers of dipterans, hymenopterans, and lepidopterans collected per sweep-day. Russian olive had the greatest number of coleopterans. When ants and chalcids were excluded from the hymenopterans, willows still had the greatest numbers and proportions of hymenopterans caught. It appears that the willow habitat is important to pollinating insects, especially bees. In contrast, saltcedar consistently had the lowest numbers and proportions of all four of the pollinator orders.

INTRODUCTION

Along the Middle Rio Grande Valley in central New Mexico, riparian gallery forests of cottonwood (*Populus fremontii*) and willow (*Salix* spp.) are being invaded by woody exotics, primarily Russian olive (*Elaeagnus angustifolia*) and saltcedar (*Tamarix* spp.). The primary cause for this invasion is the lack of pulse-flood regimes due to water diversion and flow regulation (Glenn and Nagler 2004; Katz and Shafroth 2003). These non-native plants often occur in dense thickets with many dead stems, which increase the risk of fire, thereby reducing cover of fire-sensitive native vegetation and creating a high-salinity environment detrimental to native plant re-establishment (Busch and Smith 1995; Di Tomaso 1998; Glen and Nagler 2005; Katz and Shafroth 2003; Kerpez and Smith 1987). This feed-back mechanism favors the expansion of woody exotics, particularly *Tamarix*.

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Rosemary L. Pendleton, Burton K. Pendleton, and Deborah Finch; USDA Forest Service, Rocky Mountain Research Station, Albuquerque, NM

Much effort and research has gone into exotic species eradication efforts and knowledge syntheses (for example; Di Tomaso 1998; Glenn and Nagler 2004; Horton and Campbell 1974; Katz and Shafroth 2003; Kerpez and Smith 1987; Shafroth and others 2005). One primary area of research has focused on the effects of an altered riparian ecosystem to wildlife. Several studies have focused on the impact of woody plant invasion on wildlife abundance and diversity, including birds (Brown 1990; Ellis 1995; Hunter and others 1988; Knopf and Olson 1984; Stoleson and Finch 2001), small mammals (Ellis and others 1997), amphibians and reptiles (Bateman and others 2008; Jakle and Gatz 1985; Jones 1988), and arthropods (DeLay and others 1999; Ellis and others 2000; Mund-Meyerson 1998; Watts and others 1977).

Despite considerable research on wildlife, including arthropods, few studies have examined the impact of woody exotic invasion on pollinating insects. Pollinators are indispensable contributors to biodiversity, ecosystem health, and human food production. Recent declines in pollinator abundance and health, such as catastrophic declines in honey bee populations due to Colony Collapse Disorder, have renewed interest in native pollinators and the ecosystem services they provide (Greer 1999; NAPPC). In this report, we make use of an existing dataset of insect and other arthropod collections made in the riparian zone of the Middle Rio Grande (data on file at the Forestry Sciences Laboratory, 333 Broadway SE, Albuquerque, NM) to address the following questions: 1) Does the diversity or abundance of arthropods collected from native and non-native woody plant species differ? 2) Are there differences in the insect pollinator groups associated with native and non-native species?

METHODS

Sweep nets were used to collect insects from Russian olive, saltcedar, and coyote willow (*Salix exigua*) at Bosque del Apache National Wildlife Refuge, Socorro County, NM, on multiple days throughout April and May of 1997 and 1998. These species were chosen because they flower concurrently, share similar habitats (Hoddenbach 1989) and are attractive to pollinating insects, providing both nectar and pollen (Anderson 2006; Haddock 2008; Hayes 1976; Katz and Shafroth 2003; Stevens 1989; Wiesenborn and others 2008). A direct comparison is therefore possible.

Collections were made in three habitat types growing in close proximity along the Rio Grande river near the

northern boundary of the Wildlife Refuge. Willow collections were made in two homogeneous, dense willow stringers along water conveyance structures (Low Flow, LF; South Willow, SW). Russian olive collections were made in a mixed forest of cottonwood overstory with a middle story of both Russian olive and saltcedar (North Cottonwood, NC). Saltcedar collections were made in the cottonwood forest described above, and in two saltcedar forests containing dense patches of both live and dead saltcedar interspersed with open ground (Saltcedar, SS; South Cottonwood, SC). A total of 80 collections, representing different days and/or collection sites, were made on willow, 15 on Russian olive, and 79 on saltcedar (table 1).

For each collection unit, heavy-duty sweep nets (Forestry Supplies) were swept back-and-forth over the target shrubs for a total of 10 passes. Each back-and-forth sweep was done in a different location on the plant. Sweeps were conducted on the sides of the plants at chest or head height. If only portions of the plant was flowering, the flowering areas were selected for sweeps. The majority of sweeps

were done in the late morning between 9:00 and 11:00 a.m. on days with little or no precipitation and low winds. Collected arthropods were transferred to cyanide kill-jars and transported to the laboratory, where specimens were counted and identified to family.

Arthropod numbers were summed according to plant species, site and year, then adjusted by number of collection units to account for collection effort (table 1). The adjusted abundance data were compared using the univariate option of a Multiple Response Permutation Procedure (MRPP) macro developed by Rudy King, Rocky Mountain Research Station Statistician, and based on methodology of Mielke and Berry (2001). A Bonferroni adjustment of the *P* value was made for the multiple univariate tests presented in table 2 (Miller 1981). Pairwise comparisons used the Peritz closure method (Petrondas and Gabriel 1983), which maintains Type I error at or below the specified value of 0.05. Arthropod richness was compared using the number of orders and families collected from each plant species to construct order- and family-area curves, similar to species-area curves.

Table 1—Insect collection data from Bosque del Apache National Wildlife Refuge. Data shown include collection site, year, number of collections, total number of arthropods caught at each site, and number of arthropod orders and families represented.

Species	Site	Year	No. of Collections	Total Arthropods	Arthropods per Collection	No. of Orders	No. of Families
<i>Salix exigua</i>	LF	1997	32	633	19.8	11	47
	LF	1998	24	1217	50.7	10	43
	SW	1997	24	883	36.8	11	42
<i>Elaeagnus angustifolia</i>	NC	1997	9	185	20.6	7	18
	NC	1998	6	85	14.2	6	20
	NC	1997	11	108	9.8	7	21
	NC	1998	12	52	4.3	6	16
<i>Tamarix</i> spp.	SS	1997	35	77	2.2	7	16
	SS	1998	18	260	14.4	7	23
	SC	1997	3	37	12.3	6	6

Table 2—Numbers of insects of the four major pollinator orders collected from willow, Russian olive and saltcedar at the Bosque del Apache National Wildlife Refuge during the springs of 1997 and 1998. Numbers have been corrected for collection effort. Attained significance values were based on univariate MRPP with a Bonferroni adjustment for multiple comparisons. Adjusted significance values are given in parentheses. Pairwise comparisons were computed using the Peritz closure method with an alpha of 0.05.

Arthropod Orders	<i>Salix exigua</i>	<i>Elaeagnus angustifolia</i>	<i>Tamarix</i> spp.	Attained Significance
Coleopterans per collection	1.55	4.73	1.05	0.0425 (0.255)
Dipterans per collection	11.11	5.87	1.19	0.2433
Lepidopterans per collection	0.16 a	0.00 ab	0.01 b	0.0083 (0.0498)
Hymenopterans per collection	6.81 a	3.53 ab	1.02 b	0.0052 (0.0312)
Hymenopterans minus ants and chalcidoids	0.70 a	0.20 ab	0.05 b	0.0012 (0.0072)
Other orders per collection	14.5	3.87	3.48	0.0571

In addition to the univariate results described above, MRPP was used to examine woody plant differences across the four main potential pollinating orders; *Hymenoptera* (bees, wasps, ants), *Diptera* (flies), *Lepidoptera* (butterflies, moths), and *Coleoptera* (beetles). Response variables were insect numbers of each of the four orders adjusted for collection effort and standardized in the analysis. Principal components were used to provide a visual description of group differences.

A number of sources were used to determine potential pollinator status of the different family groups. These included online sources (North American Pollinator Protection Campaign [NAPPC], Pollinator Conservation Digital Library, Montana State Extension website, U.S. Forest Service Celebrating Wildlife website), standard field guides (Borror and White 1970; White 1983), and individual articles (Franz 2007; Gilbert and Jervis 1998; Kearns 2001; Kevan and Baker 1983; Larson and others 2001; Winterton and others 2005).

RESULTS

Arthropod Abundance and Diversity

Total arthropod abundance was greatest for the willow habitat. A total of 2733 arthropods were collected from willows as compared with 270 from Russian olive and 534 from saltcedar (table 1). When corrected for collection effort, willows averaged 34.2 arthropods per collection, compared to 18.0 for Russian olive and 6.8 for saltcedar. Willow was significantly different than saltcedar based on univariate MRPP results ($p = 0.0278$). The numbers of individuals collected from each plant species are given in appendix 1. Willow collections were dominated by homopterans (cicadas, leafhoppers, aphids; 35 percent), followed by dipterans (33 percent), hymenopterans (20 percent), and coleopterans (5 percent). Russian olive collections were largely composed of dipterans (32 percent), followed by coleopterans (26 percent), hymenopterans (20 percent), and thysanopterans (thrips; 8 percent). Saltcedar collections were largely made up of hemipterans (true bugs; 37 percent), followed by dipterans (18 percent), coleopterans (16 percent), hymenopterans (15 percent), and homopterans (9 percent).

The homopterans collected were primarily leafhoppers (*Cicadellidae*), which have piercing and sucking mouthparts and feed almost exclusively on plant sap. Dipterans collected from both willow and saltcedar were largely non-biting midges (*Chironomidae*) and frit-flies (*Chloropidae*), whereas those collected from Russian olive were dance flies (*Empididae*). Frit-fly adults occur in grasses and flowers, and may serve as pollinators of small-flowered plants (for example; Neel and others 2001). Adult non-biting midges are short-lived and most do not eat. Dance flies are mostly predaceous, but are also found on

flowers and may serve as pollinators of some species (Pollinator Conservation Digital Library). Hemipterans collected from saltcedar were largely seed-feeding bugs (*Lygaeidae*). Thysanopterans, or thrips (*Thripidae*), are common plant and flower feeders that may also pollinate a few flower species such as phlox (Strakosh and Ferguson 2005). Coleopterans collected from all three woody plant species were primarily soft-winged flower beetles (*Melyridae*), with a lesser number of leaf beetles (*Chrysomelidae*) and false flower beetles (*Scrapyidae*). Soft-winged flower beetles, in particular, constitute an important pollinator group (Mawdsley 2003). Hymenopteran collections were dominated by ants (*Formicidae*), followed by chalcidoids (*Chalcidoidea*). Chalcidoids are a large group of tiny wasps whose larvae are parasitic on other insects (Borror and White 1970). Adults feed on high-carbohydrate energy sources such as nectar and honeydew, as do other parasitoid wasps (Felder and Landis 2007).

Willows had the greatest number of arthropod orders and families represented. Fourteen orders and 72 families were collected from willow; seven orders and 30 families were collected from Russian olive; and 10 orders and 46 families were collected from saltcedar. Order- and species-area curves show higher accumulation rates for willow as compared with saltcedar (figure 1). Russian olive had too few collection units for a meaningful comparison.

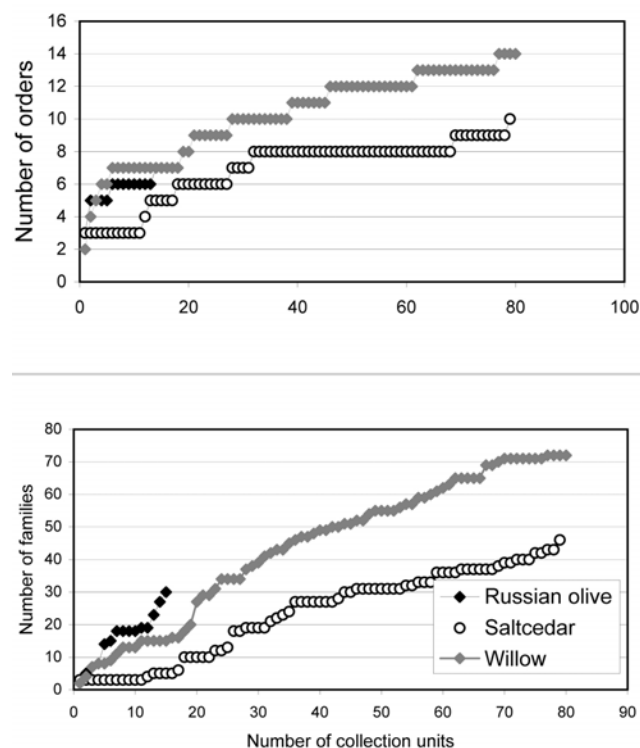


Figure 1—Order- and family-area curves for arthropods collected from willow, Russian olive, and saltcedar.

Potential Pollinators

Results of the multivariate MRPP on the four primary insect pollinator orders showed a significant difference in the potential pollinator assemblage of willow versus saltcedar ($p = 0.0008$). Willow had significantly greater numbers of both lepidopterans and hymenopterans per collection, even when ants and chalcidoids (the largest groups) were excluded from the hymenopterans (table 2). Russian olive had the greatest number of coleopterans collected, although this was only marginally significant.

The difference between the insect assemblage of willow as opposed to the two exotic species is depicted through principal components in figure 2. The first component, representing 53 percent of the variation, is a comparison of coleopterans against all other insect orders collected (table 3). The second component, comprising 26 percent of the variation, contrasts coleopterans and, to a lesser extent hymenopterans, with other non-pollinator orders. Relatively speaking, willow had fewer coleopterans whereas Russian olive had high numbers. The 1997 saltcedar collection from the NC site was similar to Russian olive in having relatively high numbers of coleopterans. These two species co-occurred at this site, which consisted of a cottonwood overstory with a mixed exotic middle story.

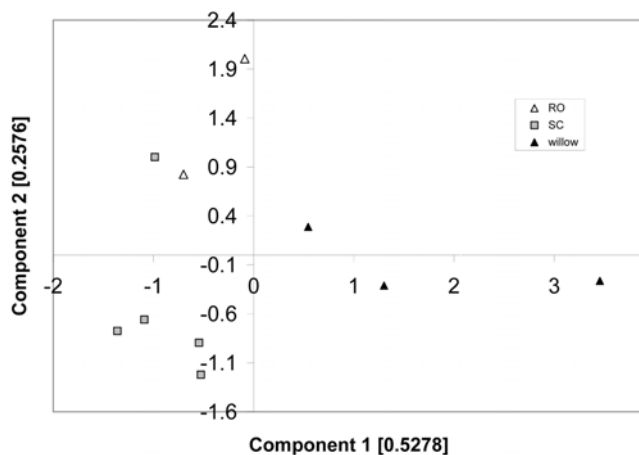


Figure 2—Results from principal components depicting differences in insect assemblages collected from willow, Russian olive and saltcedar.

Soft-winged flower beetles comprised the bulk of coleopterans collected, however a number of other beetle families were also collected. Six beetle families were collected from Russian olive, seven from saltcedar, and ten from willow (appendix 1). All families collected, with the exception of ladybird beetles (*Coccinellidae*) and fireflies (*Lampyridae*), have at least some species commonly found on flowers (Borror and White 1970) or implicated in pollination (Pollinator Conservation Digital Library). Therefore, we considered beetles in the remaining 11 families to be potential pollinators. The preponderance of soft-winged flower beetles makes them particularly

important as potential pollinators, especially for Russian olive and saltcedar (figure 3A).

Lepidopterans comprise the second largest order of insects, but very few were collected in this study. Those collected were either geometer moths (*Geometridae*), noctuid moths (*Noctuidae*), or unidentified caterpillars (appendix 1). All nine adults of the two moth groups can be considered potential pollinators, and these were found exclusively on willow.

Flies are among the most frequent insect visitors to flowers (Kearns 2001) and are more important as pollinators than is commonly recognized. We collected ten families of flies from Russian olive, 15 from saltcedar, and 22 from willow (appendix 1). We found reports of nectar consumption and/or flower visitation for nearly all fly families collected in this study. The single exception was picture-winged flies, *Otitidae* (now *Ulidiidae*), for which we could find no information on adult feeding behavior. Non-biting midges comprised the bulk of dipterans collected from willow (figure 3B). Most adult non-biting midges (*Chironomidae*) do not eat. However, the family is listed as nectar feeders on arctic plants, including willows, in several sources (Gilbert and Jervis 1998; Kevan and Baker 1983; Larson and others 2001). Consequently, we have included all flies as potential pollinators, although the non-biting midges are questionable. Dance flies comprised the bulk of dipterans collected from Russian olive, whereas saltcedar hosted midges, frit-flies, and a smattering of other families (figure 3B).

Hymenopterans consist of ants, wasps and bees. Four families were collected from Russian olive, four from saltcedar, and ten from willow (appendix 1). The vast majority were ants and parasitoid wasps. Although a few specialized cases of ant pollination have been documented, ants are rarely effective as pollinators because antibiotic body excretions render pollen inviable (Wagner 2000). Parasitic (chalcidoid, ichneumon, braconid) and predaceous (sphecid, vespid) wasps are known to pollinate, as adults feed on rich sugar sources, including floral nectaries. Parasitic wasps, especially chalcidoids, made up the bulk of hymenopterans collected from all three species (figure 3C). Bees, however, are the most important and effective of all pollinators. Their mouthparts and bodies are well adapted to make use of floral resources (Kevan and Baker 1983), which they use both as food and to provision their nests. The total number of bees collected in this study was relatively small, possibly due to their quickness in flight. Twenty-eight bees in four families (*Andrenidae*, *Apidae*, *Colletidae*, and *Halictidae*) were collected from willow, compared with two bees in one family (*Halictidae*) from saltcedar. No bees were collected from Russian olive. The proportions of potential pollinators by order are shown in figure 4. Dipterans are the most abundant visitors for all

three plant species. In willow, most of these are non-biting midges. If midges are excluded, hymenopterans and dipterans are about equal in abundance on willow, whereas dipterans and coleopterans make up the bulk of visitors on Russian olive and saltcedar.

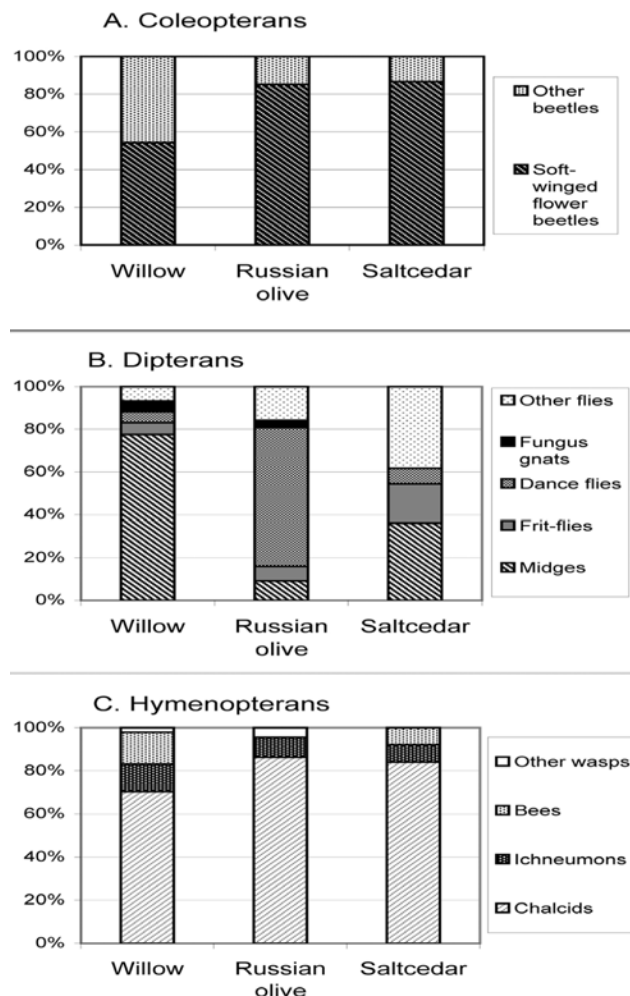


Figure 3--Family make-up of three of the four principal pollinating insect orders by woody plant species. Lepidopterans are not shown because adults were collected only from willow.

DISCUSSION

Arthropods

In this study, we found willows to have a greater abundance and diversity of arthropods than did either saltcedar or Russian olive. Findings from previous studies on arthropod communities of exotic and native vegetation have been mixed. Shafroth and others (2005; p. 236) cite several Master's theses (Liesner 1971; Miner 1989; Stevens 1985) that found saltcedar to be depauperate in arthropod assemblages when compared with native vegetation. DeLay and others (1999) found spring arthropod collections from

saltcedar habitat to be significantly lower in abundance than collections from either cottonwood or willow habitats. Cohan and others (1978) reported low use of saltcedar stands by insectivorous bird species, which correlated with their finding of significantly less insect biomass in saltcedar as compared with other riparian vegetation types, including willow. Other researchers have also reported greater avian density and diversity in cottonwood-willow communities than in saltcedar (see Kerpez and Smith 1987). Arthropod abundance is a better predictor of insectivorous bird density than either foliage height or volume (Brush and Stiles 1986).

Existing studies of insects and insectivorous birds on Russian olive show a similar pattern to that depicted above for saltcedar. Waring and Tremble (1993) studied insect herbivores of native and exotic woody species along the San Juan River in Utah. They found richness and density of insect herbivores to be lowest on Russian olive and highest on coyote willow. Brown (1990) found willow sites to have higher bird species richness, relative abundance, and density, and more foraging guilds and nesting guilds than Russian olive sites. All foraging guilds (herbivore, insectivore, granivore, omnivore) preferred willow over Russian olive. She attributed differences in bird use to the relatively low abundance of insects found in Russian olive stands. Knopf and Olson (1984) also reported a lower richness and diversity of bird species in Russian olive sites than in native riparian communities.

In contrast, Ellis (1995) reported that insectivorous birds readily used saltcedar for foraging. Both Ellis and others (2000) and Mund-Meyerson (1998) concluded that saltcedar communities often have diverse and abundant arthropod assemblages. Tepedino and Griswold, working in Grand Staircase-Escalante National Monument recorded a total of 210 species of bees occurring on tamarisk (Tery Griswold, personal communication 3/20/09). Ellis and others (2000) reported a greater abundance of surface-dwelling arthropods under cottonwood than saltcedar, but attributed this solely to the high number of exotic isopods caught in the cottonwood habitat. Mund-Meyerson (1998) found no difference in either abundance or diversity of arboreal arthropod communities of cottonwood, saltcedar, and Russian olive. Shafroth and others, in their 2005 review paper, state that the abundances of several common insect families may be comparable or greater on saltcedar than on most natives (Anderson and others 2004; as cited in Shafroth and others 2005). Carothers and others (1976) reported higher arthropod abundance but lower diversity for spring collections on saltcedar compared with willow. In this case, high numbers of thrips contributed to the overall abundance. Stevens (1989) commented on the "astonishing abundance" of thrips on saltcedar flowers.

Table 3—Loadings for eigenvectors from principal components analysis.

Response	Component				
	1	2	3	4	5
Coleopterans	-0.1022	0.9337	-0.2313	0.2239	0.1185
Dipterans	0.5832	0.0534	0.5715	0.5551	0.1493
Hymenopterans	0.6305	0.2766	0.0456	-0.6506	-0.3171
Lepidopterans	0.1186	-0.0261	-0.0210	-0.3502	0.9285
Other orders	0.4877	-0.2193	-0.7857	0.3095	0.0304

What might be reasons for the above disparity in results? Most comparison studies of arthropods in exotic and native riparian vegetation have been done in the context of a food source for insectivorous fauna. As such, they have compared arthropod abundance on a community basis. Local plant species composition is a better predictor of arthropod assemblages than either vegetation structure or environmental conditions (Schaffers and others 2008). Cottonwood areas may or may not retain a mixed native mid-story containing willows. The presence or absence of herbaceous understory plants may also influence results. Our data directly compare the two exotic plants with a native willow species. Willows are known to have one of the richest and most diverse insect faunas of all tree families (Southwood 1961). Most of the previous reports did not specifically examine the importance of willows.

The second reason involves a time lag as insects and other arthropod species adapt to a novel food source. Southwood (1961) documented associations between insect assemblages and various tree species. He determined that the number of insect species associated with a tree species was influenced by both the amount of time the plant has been in a given area (introduction history) and its general abundance or scarcity within that region. It is therefore likely that the numbers of insect species associated with saltcedar and Russian olive have increased since their introduction into the United States. It would be interesting to compare insect assemblages in areas of recent expansion with those areas of the first early introductions.

Pollinators

While several studies have examined the effects of exotic woody introductions on arthropods in general, little attention has been given to their effects on pollinating insects. We know of only a few papers on saltcedar (Nelson and Andersen 1999; Tepedino and others 2008; Wiesenborn and others 2008) and none on Russian olive. Interest in pollinator conservation has grown in recent years as our understanding of their importance has grown. It is estimated that over 67 percent of flowering plants, including many crop species, depend on insects for pollination (Tepedino 1979). Current threats to native pollinators include excessive use of pesticides, habitat fragmentation, habitat loss, and introductions of exotic plant and animal species

(Kearns and Inouye 1997). Inclusion of a variety of forb and flowering shrub species in revegetation efforts is worth consideration.

As previously indicated, the majority of pollinators are found in one of four main insect orders; Coleoptera (beetles), Diptera (flies), Lepidoptera (butterflies and moths), and Hymenoptera (bees, wasps and ants). Beetles make up the largest group of pollinating insects and are considered to be the most primitive (Kevan and Baker 1983). Pollination occurs as they move from flower to flower, chewing their way through the floral parts, becoming covered with pollen. This is often referred to as “mess and soil” pollination. Many are also considered pests of economic importance, as they damage the flowers and plants upon which they feed. Mawdsley (2003) identified 11 beetle families as important pollinating groups, seven of which were collected in this study. The *Buprestidae*, *Curculionidae*, *Mordellidae*, *Scarabaeidae*, and *Staphylinidae* were poorly represented in our study, but the *Chrysomelidae* (leaf beetles) and *Melyridae* (soft-winged flower beetles) were the two most common families collected. The flower beetles were common on all three woody plants and have been reported as pollinators of both saltcedar and willow species (Mawdsley 2003). Leaf beetles were primarily collected from willow, suggesting that willow species may be important in the conservation of this family of pollinators. Several leaf beetle species are herbivore specialists on willow leaves (Sandra Brantley, personal communication).

Flies are the second most important order of flower visitors (Larson and others 2001). At least 71 families of flies (dipterans) are known to contain common visitors to flowers (Kearns 2001). These range from opportunistic feeders to specialized nectarivores and vary significantly in their effectiveness as pollinators. Important flower-visiting families include the larger *Bombyliidae*, *Syrphidae*, *Anthomyiidae*, *Tachinidae*, *Calliphoridae*, as well as the smaller *Bibionidae*, *Chironomidae*, *Culicidae*, *Empididae*, *Mycetophilidae*, and *Cecidomyiidae* (Kearns 2001). Most of the flies collected in this study were small flies of the families *Chironomidae*, *Empididae*, *Mycetophilidae*, and *Chloropidae* that feed on exposed nectaries. These small flies are generally considered to be inefficient pollinators,

as their small size precludes carrying heavy pollen loads. Nevertheless, numerous visits to flowers by small inefficient pollinators may still result in significant pollination, especially in the absence of more efficient pollinators. We collected few large flies of the families indicated above. Those that were collected came from either saltcedar or willow. As yet, our understanding of flies and fly pollination is limited. Far more research on fly pollination, community composition, and conservation status is needed (Kearns 2001).

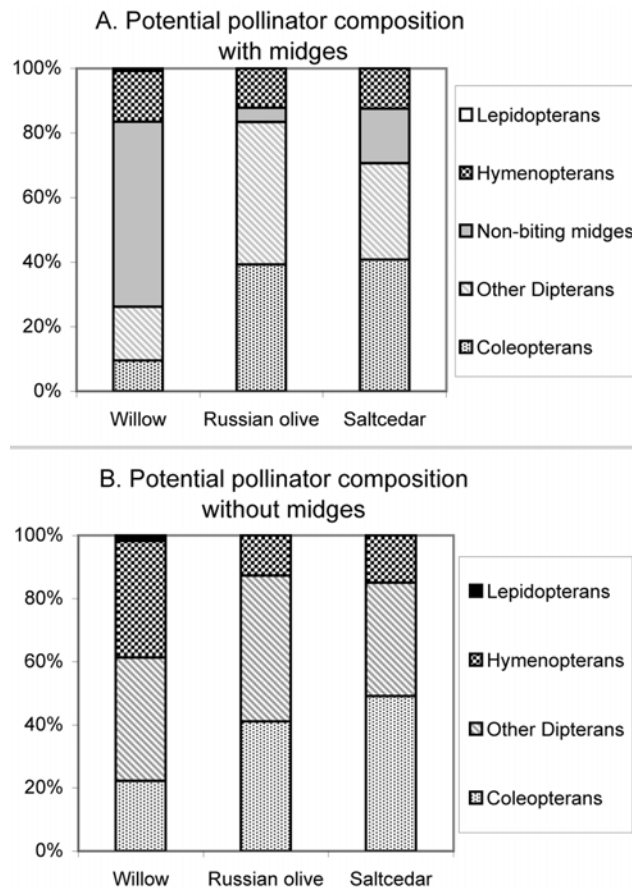


Figure 4—The proportions of potential pollinators by order for willow, Russian olive and saltcedar when non-biting midges are included (A) and excluded (B).

Adult butterflies and moths feed extensively on nectar and pollen (Kevan and Baker 1983), and so have an intimate relationship with flowers. They have been proposed as ecological indicators of riparian quality in the western United States (Nelson 2007). In this study, we collected few moths and no butterflies. What moths we did collect were all from willow and belonged to the families *Geometridae* and *Noctuidae*. Geometers and noctuids, along with sphinx moths, are considered to be the most efficient of moth pollinators (Montana State University Extension website). Nelson and Andersen (1999) found that saltcedar and restored riparian habitats along the lower Colorado River

lacked the nectar and larval resources needed for butterflies historically found in cottonwood/willow vegetation. These findings suggest that willows may be an important plant resource for lepidopteran pollinators.

Hymenopterans are the most important order of flower visitors (Kevan and Baker 1983; Tepedino 1979). Parasitic wasps were the most common group of potential pollinators collected. They appear to be generalized flower visitors and were found on all three woody plant species (Kevan and Baker 1983). Bees, however, are the most efficient and important of all pollinator groups. Their life histories are intimately linked with those of flowering plants, as they depend completely on floral resources. Both generalist and specialist foragers exist. It is estimated that less than two-thirds of bee species have been described (Kearns and Inouye 1997). Other than two halictids collected on saltcedar, all of the bees and bee families collected in this study occurred on willow, indicating its importance as a floral resource, both for nectar and the copious amounts of pollen produced.

Willows are primarily insect pollinated but wind may also play a role, depending on the species (Sacchi and Price 1988). Willows have one of the highest numbers of insect species occurring on trees, even when the effects of introduction history and geographic abundance are removed (Southwood 1961). A large number of insect species have been identified as floral visitors or pollinators of willow (Kevan and Baker 1983; Pollinator Conservation Digital Library website). Our results support these observations. Willows supported the greatest diversity and abundance of arthropods, potential pollinators, and bees.

Little is known of the pollination needs of woody exotics. Stevens (1989) reported that saltcedar requires insect pollinators to successfully produce seed. The most commonly mentioned pollinators are honeybees (Shrader 1978). Wiesenborn and others (2008) collected a total of 17 families and four orders of floral-visiting insects from saltcedar flowers in Arizona. Relative pollen loads indicated that honey bees were the most important pollinators. A number of native bees of the *Apidae*, *Halictidae*, and *Megachilidae* also carried pollen, but were found in low numbers. The authors also indicated that syrphid flies may be important pollinators due to the high frequency of visits. Our data suggest that small flies and flower beetles may supply the majority of saltcedar pollination in our study area, as honeybees are relatively rare.

The pollination needs of Russian olive in North America are virtually unknown, although the consensus is that they are insect pollinated (Olson and Barbour 2008). As with saltcedar, the majority of potential pollinators we collected were flower beetles and small flies. No lepidopterans or

bees were collected, although Russian olive has been promoted for the production of honey (Hayes 1976). Clearly, pollination studies are needed for this species.

The value of woody exotics to major arthropod communities in general, and to pollinator groups in particular, is largely relative and depends on the associated plant species. While we demonstrated that replacement of willow understory by woody exotics has a detrimental effect on the pollinator community, especially bees, this may not be true for all plant/pollinator associations. The presence of an herbaceous layer can have a profound effect on the abundance and richness of floral visiting insects (Campbell and others 2007). In addition to food for adults, insects require suitable larval hosts and nesting sites to complete their life cycle. Wiesenborn and others (2008) attributed the lack of butterflies to a lack of larval host plants. They also observed that saltcedar did not readily form hollow stems for cavity nesting bees and wasps. Parasitic wasps require other insects as larval food sources, and saltcedar supports a low diversity of insect herbivores (Wiesenborn and others 2008).

Nevertheless, the effects of exotics on insect populations are not all negative. The addition of saltcedar in mixed stands may, in some cases, actually increase the abundance of native bees. Again, this depends on the structure and composition of associated vegetation. Exotic species, such as tamarisk, may be most attractive when native species are not available or are not in flower (Terry Griswold, personal communication 3/20/09). Tepedino and others (2008) compared bee assemblages visiting exotic and native flowering plants in Capitol Reef National Park. They found that exotic species, including saltcedar, were host to a number of native generalist bee species and concluded that exotics could potentially increase the carrying capacity and population size of generalist bees. Saltcedar provides an early source of pollen for overwintering bees (Kerpez and Smith 1987) and may also be a refugia for bees when nearby agricultural fields are being sprayed (Horton and Campbell 1974).

CONCLUSIONS

This study highlights the importance of maintaining willows in the riparian community. When compared head-to-head with woody invasives, willows appeared to be more important to pollinating insects, especially bees. Approximately 73 percent of the world's cultivated crops are pollinated by some species of bee (Food and Agriculture Organization of the United Nations 2004). The effect of woody exotics on native bee populations is therefore of considerable importance. Cohan and others (1978) suggested that a reduction in density of pure saltcedar stands and subsequent replacement with mesquite and willow would be of benefit to wildlife. This report

suggests additional benefits to native pollinators, especially bees, of maintaining or restoring willow species in southwestern riparian zones.

ACKNOWLEDGEMENTS

Thanks are extended to Dave Hawksworth, Linda DeLay, and others who spent countless hours in the field and lab collecting, sorting and identifying the arthropods used in this analysis. We also wish to thank reviewers Sandra Brantley and Terry Griswold for comments and suggestions that greatly improved the manuscript.

REFERENCES

- Anderson, M. 2006. *Salix exigua*. In: Fire Effects Information System. Online. USDA, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). Available: www.fs.fed.us/database/feis/ [2008, December 2].
- Anderson, B.W., Russell, P.E., Ohmart, R.D. 2004. Riparian revegetation: an account of two decades of experience in the arid southwest. Blyth, CA: Avvar Books.
- Bateman, H.L., Chung-MacCoubrey, A., Snell, H.L. 2008. Impact of non-native plant removal on lizards in riparian habitats in the southwestern United States. *Restoration Ecology*. 16: 180-190.
- Borror, D.J., White, R.E. 1970. A field guide to insects. Boston, MA: Houghton Mifflin Company. 404 p.
- Brown, C.R. 1990. Avian use of native and exotic riparian habitats on the Snake River, Idaho. MS Thesis. Ft. Collins, CO: Colorado State University.
- Brush, T., Stiles, E.W. 1986. Using food abundance to predict habitat use by birds. In: Verner, J., Morrison, M.L., Ralph, C.J., eds. *Wildlife 2000, modeling habitat relationships of terrestrial vertebrates*. Madison, WI: University of Wisconsin Press: 57-63.
- Busch, D.E., Smith S.D. 1995. Mechanisms associated with decline of woody species in riparian ecosystems of the southwestern U.S. *Ecological Monographs*. 65: 347-370.
- Campbell, J.W., Hanula, J.L., Waldrop, T.A. 2007. Effects of prescribed fire and fire surrogates on floral visiting insects of the blue ridge province in North Carolina. *Biological Conservation*. 134: 393-404.
- Carothers, S.W., Aitchison, S.E., Marpiscak, M.M., Ruffner, G.A., Sharber, J.J. 1976. An ecological survey of the riparian zone of the Colorado River between Lee's Ferry and Grand Wash Cliffs, Arizona. Colorado River Res. Serv. Tech. Rep. 10. USDI, National Park Service. 251 p.
- Cohan, D.R., Anderson, B.W., Ohmart, R.D. 1978. Avian population responses to salt cedar along the lower colorado River. In: Johnson, R.R., McCormick, J.F., eds. *Strategies for protection and management of floodplain wetlands and other riparian ecosystems*. General Technical Report WO-12. Washington, DC: USDA, Forest Service: 371-382.
- DeLay, L., Finch, D.M., Brantley, S., Fagerlund, R., Means, M.D., Kelly, J.F. 1999. Arthropods of native and exotic vegetation and their association with willow flycatchers and Wilson's warblers. In: Finch, D.M., Whitney, J.C., Kelly, J.F., Loftin, S.R., comps. *Rio Grande ecosystems: linking land, water, and people. Toward a sustainable future for the Middle Rio Grande Basin*. 1998 June 2-5, Albuquerque, NM. Proc. RMRS-P-7. Ogden, UT: USDA Forest Service, Rocky Mountain Research Station: 216-221.
- Di Tomaso, J.M. 1998. Impact, biology, and ecology of saltcedar (*Tamarix* spp.) in the southwestern United States. *Weed Technology*. 12: 326-336.
- Ellis, L.M. 1995. Bird use of saltcedar and cottonwood vegetation in the Middle Rio Grande Valley of New Mexico, U.S.A. *Journal of Arid Environments*. 30: 339-349.
- Ellis, L.M., Crawford, C.S., Molles, M.C., Jr. 1997. Rodent communities in native and exotic riparian vegetation in the Middle Rio Grande Valley of central New Mexico. *Southwestern Naturalist*. 42: 13-19.
- Ellis, L.M., Molles, M.C., Jr., Crawford, C.S., Heinzelmann, F. 2000. Surface-active arthropod communities in native and exotic riparian vegetation in the Middle Rio Grande Valley, New Mexico. *Southwestern Naturalist*. 45: 456-471.

- Fiedler, A.K., Landis, D.A. 2007. Attractiveness of Michigan native plants to arthropod natural enemies and herbivores. *Environmental Entomology*. 36: 751-765.
- Food and Agriculture Organization of the United Nations. 2004. Conservation and management of pollinators for sustainable agriculture – the international response. In: Freitas, B.M., Pereira, J.O.P., eds. Solitary bees – conservation, rearing and management for pollination. Fortaleza: Imprensa Universitária: 19-25.
- Franz, N.M. 2007. Pollination of *Anthurium* (Araceae) by derelomine flower weevils (Coleoptera: Curculionidae). *Revista de Biología Tropical*. 55:269-277.
- Gilbert, F., Jervis, M. 1998. Functional, evolutionary and ecological aspects of feeding-related mouthpart specializations in parasitoid flies. *Biological Journal of the Linnean Society*. 63:495-535.
- Glenn, E.P., Nagler, P.L. 2004. Comparative ecophysiology of *Tamarix ramosissima* and native trees in western U.S. riparian zones. *Journal of Arid Environments*. 61: 419-446.
- Greer, L. 1999. Alternative pollinators: native bees. *Appropriate Technology Transfer for Rural Areas*. Publication IP126. Fayetteville, AR. 14 p.
- Hayes, B. 1976. Planting the *Elaeagnus* Russian and autumn olive for nectar. *American Bee Journal*. 116:74,82.
- Haddock, M. 2008. Sandbar willow. *Kansas Wildflowers and Grasses*. Online. Available: <http://www.kswildflower.org/details.php?flowerID=426>. 2008, December 2.
- Hoddenbach, G. 1989. Tamarix control. In: Kunzmann, M. R., Johnson, R. Roy, Bennett, Peter, tech. coords. Tamarisk control in southwestern US: Proceedings, 1987 September 2-3, Tucson, AZ. Special Report No. 9. Tucson, AZ: National Park Service, Cooperative National Park Resources Studies Unit, School of Renewable Natural Resources: 116-125.
- Horton, J.S., Campbell, C.J. 1974. Management of phreatophyte and riparian vegetation for maximum multiple use values. Res. Pap. RM-117. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Forest and Range Experiment Station. 23 p.
- Hunter, W.C., Ohmart, R.D., Anderson, B.W. 1988. Use of exotic saltcedar (*Tamarix chinensis*) birds in arid riparian systems. *The Condor*. 90:113-123.
- Jakle, M.D., Gatz, T.A. 1985. Hepetofaunal use of four habitats of the Middle Gila River drainage, Arizona. In: Johnson, R.R., Ziebell, C.D., Patton, D.R., Ffolliott, P.F., Hamre, R.H., coords. Riparian ecosystems and their management: reconciling conflicting uses. First North American Riparian Conference, 16-18 April 1985, Tucson, AZ. General Technical Report RM-166. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station: 222-227.
- Jones, K.B. 1988. Comparison of herpetofaunas of a natural and altered riparian ecosystem. In: Szaro, R.C., Severson, K.E., Patton, D.R., eds. Management of amphibians, reptiles, and small mammals in North America. Proceedings of a symposium, 19-21 July 1988, Flagstaff, AZ. General Technical Report RM-166. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station: 222-227.
- Katz, G.L., Shafroth, P.B. 2003. Biology, ecology and management of *Elaeagnus angustifolia* L. (Russian olive) in western North America. *Wetlands*. 23: 763-777.
- Kearns, C.A. 2001. North American dipteran pollinators: assessing their value and conservation status. *Conservation Ecology* 5: 5. Online. Available: <http://www.consecol.org/vol5/iss1/art5/>. January 8, 2009.
- Kearns, C.A., Inouye, D.W. 1997. Pollinators, flowering plants, and conservation biology. *BioScience*. 47: 297-307.
- Kerpez, T.A., Smith, N.S. 1987. Saltcedar control for wildlife habitat improvement in the southwestern United States. Resource Publication 169. Washington, DC: USDI, Fish and Wildlife Service. 14 pp.
- Kevan, P.G., Baker, H.G. 1983. Insects as flower visitors and pollinators. *Annual Review of Entomology*. 28: 407-453.
- Knopf, F.L., Olson, T.T. 1984. Naturalization of Russian-olive: implications to Rocky Mountain wildlife. *Wildlife Society Bulletin*. 12: 289-298.
- Larson, B.M.H., Kevan, P.G., Inouye, D.W. 2001. Flies and flowers: taxonomic diversity of anthophiles and pollinators. *The Canadian Entomologist*. 133: 439-465.
- Mawdsley, J.R. 2003. The importance of species of *Dasytinae* (Coleoptera: Melyridae) as pollinators in western North America. *The Coleopterists' Bulletin*. 57: 154-160.
- Mielke, P.W. Jr., Berry, K.J. 2001. Permutation Methods: A Distance Function Approach. New York: Springer-Verlag. 352 p.
- Miller, R.G. Jr. 1981. Simultaneous Statistical Inference, 2nd Ed. New York:Springer-Verlag. 299p.
- Montana State University Extension. Pollinating insects. Online. Available: <http://diagnostics.montana.edu/Insect/Pollinators/>. January 9, 2009.
- Mund-Meyerson, M.J. 1997. Arthropod abundance and composition on native vs. exotic vegetation in the Middle Rio Grande riparian forest as related to avian foraging. M.S. Thesis. Albuquerque, NM: University of New Mexico. 48 pp.
- NAPPC. North American Pollinator Protection Campaign. Online. Available: <http://www.nappe.org/>. December 6, 2008.
- Neel, M.E., Ross-Ibarra, J., Ellstrand, N.C. 2001. Implications of mating patterns for conservation of the endangered plant *Eriogonum ovalifolium* var. *vineum* (Polygonaceae). *American Journal of Botany*. 88: 1214-1222.
- Nelson, S.M. 2007. Butterflies (Papilionoidea and Hesperioidea) as potential ecological indicators of riparian quality in the semi-arid western United States. *Ecological Indicators*. 7: 469-480.
- Nelson, S.M., Andersen, D.C. 1999. Butterfly (Papilionoidea and Hesperioidea) assemblages associated with natural, exotic, and restored riparian habitats along the lower Colorado River, USA. *Regulated Rivers: Research and Management*. 15: 485-504.
- Olson, D.F., Jr., Barbour, J.R. 2008. *Elaeagnus* L. In: Bonner, F.T., Karrfalt, R.P., eds. The woody plant seed manual. Agriculture Handbook 727. Washington, DC: USDA Forest Service: 484-487.
- Petrondas, D.A., Gabriel, K.R. 1983. Multiple Comparisons by Rerandomization Tests. *Journal of the American Statistical Association*. 78(384): 949-957.
- Pollinator Conservation Digital Library. Online] Available: libraryportals.com/PCDL. January 5, 2009.
- Sacchi, C.F., Price, P.W. 1988. Pollination of the arroyo willow, *Salix lasiolepis*: role of insects and wind. *American Journal of Botany*. 75: 1387-1393.
- Schaffers, A.P., Raemakers, I.P., Sykora, K.V., ter Braak, C.J.F. 2008. Arthropod assemblages are best predicted by plant species composition. *Ecology*. 89: 782-794.
- Shafroth, P.B., Cleverly, J.R., Dudley, R.L., Taylor, J.P., van Riper, C., III, Weeks, E.P., Stuart, J.N. 2005. Control of Tamarix in the western United States: Implications for water salvage, wildlife use, and riparian restoration. *Environmental Management*. 35: 231-246.
- Shrader, T.H. 1978. Selective management of phreatophytes for improved utilizations of natural flood-plain resources. In: Johnson, R.R., McCormick, J.F., tech. coords. Strategies for protection and management of floodplain wetlands and other riparian ecosystems. General Technical Report WO-12. Washington, DC: USDA, Forest Service: 16-44.
- Southwood, T.R.E. 1961. The numbers of species of insect associated with various trees. *Journal of Animal Ecology*. 30: 1-8.
- Stevens, L.E. 1989. The status of ecological research on tamarisk (Tamaricaceae: *Tamarix ramosissima*) in Arizona. In: Kunzmann, M.R., Johnson, R.R., Bennett, P., tech. coords. Tamarisk control in southwestern United States: Proceedings, 1987 September 2-3, Tucson, AZ. Special Report No. 9. Tucson, AZ: National Park Service, Coop National Park Resources Studies Unit, School of Renewable Natural Resources: 99-105.
- Stoleson, S.H., Finch, D.M. 2001. Breeding bird use of and nesting success in exotic Russian olive in New Mexico. *Wilson Bulletin*. 113: 452-455.
- Strakosh, S.C., Ferguson, C.J. 2005. Pollination biology of four southwestern species of *Phlox* (Polemoniaceae). *Southwestern Naturalist*. 50: 291-301.
- Tepedino, V.J. 1979. The importance of bees and other insect pollinators in maintaining floral species composition. *Great Basin Naturalist Memoirs* 3: 39-150.
- Tepedino, V.J., Bradley, B.A., Griswold, T.L. 2008. Might flowers of invasive plants increase native bee carrying capacity? Intimations from Capitol Reef National Park, Utah. *Natural Areas Journal*. 28: 44-50.
- U.S. Forest Service. Celebrating Wildflowers. Online. Available: <http://www.fs.fed.us/wildflowers/index.shtml>. January 9, 2009.
- Wagner, D. 2000. Pollen viability reduction as a potential cost of ant association for *Acacia constricta* (Fabaceae). *American Journal of Botany*. 87: 711-715.
- Waring, G.L., Tremble, M. 1993. The impact of exotic plants on faunal diversity along a Southwestern river. Unpublished Report to: Nature Conservancy, Arlington VA. Contract #B0767000-910524. 33 p.
- Watts, J.G., Liesner, D.R., Lindsey, D.L. 1977. Salt cedar--a potential target for biological control. *Bulletin* 650. Las Cruces, NM: New Mexico State University, Agricultural Experiment Station. 28 p.
- White, R.E. 1983. A field guide to the beetles of North America. Boston, MA: Houghton Mifflin Co. 368 pp.

Wiesenborn, W.D., Heydon, S.L., Lorenzen, K. 2008. Pollen loads on adult insects from Tamarisk flowers and inferences about larval habitats at Topock Marsh, Arizona. *Journal of the Kansas Entomological Society*. 81(1): 50–60.

Winterton, S.L., Skevington, J.H., Lambkin, C. L. 2005. Stiletto-flies of Australasia. Online. Available: http://www.cdffa.ca.gov/phpps/ppd/Lucis/Therevidae/Austherevid/key/Austherevid/Media/Html/opening_page.html. January 9, 2009.

APPENDIX 1—Numbers of arthropods by family that were collected from each of the three woody plant species.

Order	Family	Common name	Russian olive	Saltcedar	Willow
Araneida	Araneidae	Orb-weaving spiders	5	11	15
	Anyphaenidae	Anyphaenid sac spiders		2	
	Clubionidae	Sac spiders			2
	Lycosidae	Wolf spiders			1
	Mimetidae	Pirate spiders			1
	Oxyopidae	Lynx spiders	1		2
	Philodromidae	Running spiders	1	1	5
	Salticidae	Jumping spiders	3	2	4
	Tetragnathidae	Thick-jawed orb weavers			1
	Thomisidae	Crab spiders	2	3	53
	Unknown		2	4	11
Coleoptera	Bruchidae	Seed beetles	3		
	Buprestidae	Metallic wood-borers		1	
	Chrysomelidae	Leaf beetles	2	8	34
	Cleridae	Checkered beetles			3
	Coccinellidae	Ladybird beetles		1	9
	Curculionidae	Snout beetles		1	5
	Dermestidae	Dermestid beetles	2		4
	Lampyridae	Fireflies			1
	Melyridae	Soft-winged flower beetles	52	63	57
	Mordellidae	Tumbling flower beetles		1	
	Scaptiidae	False flower beetles	11	8	9
	Scarabaeidae	scarab beetles			1
	Staphylinidae	Rove beetles	1		1
Collembola	Entomobryidae	Springtails		1	2
	Sminthuridae	Globular springtails		2	
Dermoptera	Unknown	Earwigs			1
Diptera	Asilidae	Robber flies			1
	Anthomyiidae	Anthomyiid flies			1
	Bibionidae	March flies	1		7
	Bombyliidae	Bee flies		3	5
	Calliphoridae	Blow flies			1
Diptera	Cecidomyiidae	Gall Gnats			3
	Ceratopogonidae	Punkies, biting midges	2		2
	Chironomidae	Midges	8	34	689
	Chloropidae	Frit flies	6	17	50
	Culicidae	Mosquitos		4	1
	Dolichopodidae	Long-legged flies	6	3	3
	Empididae	Dance flies	57	7	45
	Muscidae	House flies		1	2
	Mycetophilidae	Fungus gnats	3		44
	Otitidae	Picture-winged flies	1	1	6
	Phoridae	Hump-backed flies		10	8
	Pipunculidae	Big-headed flies			1
	Sciaridae	Dark-winged fungus gnats	2		3
	Sciomyzidae	Marsh flies		2	
	Syrphidae	Syrphid flies		1	

APPENDIX 1 (continued) —Numbers of arthropods by family collected from each of the three woody plant species.					
Order	Family	Common name	Russian olive	Saltcedar	Willow
Diptera	Tachinidae	Tachnid flies		1	1
	Therevidae	Stiletto flies		2	1
	Tipulidae	Crane flies	2	4	6
	Trioxoscelididae	Trioxoscelidid flies		4	9
Hemiptera	Berytidae	Stilt bugs		5	
	Corimelaenidae	Negro bugs			1
	Lygaeidae	Seed bugs	18	149	33
	Miridae	Leaf or plant bugs		37	8
	Phymatidae	Ambush bugs			1
	Reduviidae	Assassin bugs		1	
	Thyreocoridae	Negro bugs		1	
	Tingidae	Lace bugs		6	5
Homoptera	Aphidae	Aphids	3	16	77
	Cicadellidae	Leafhoppers	2	30	837
	Dictyopharidae	Dictyopharid planthoppers		1	14
	Psyllidae	Psyllids			3
	Unknown				23
Lepidoptera	Geometridae	Geometer moths			8
	Noctuidae	Noctuid moths			1
	Unknown	caterpillars		1	4
Hymenoptera	Andrenidae	Andrenid bees			7
	Apidae	Bumble, honey bees, etc.			7
	Braconidae	Braconids		2	17
	Chalcidoidea	Unknown chalcidoids	19	21	133
	Colletidae	Yellow-faced bees			3
	Formicidae	Ants	31	56	356
	Halictidae	Sweat bees		2	11
	Ichneumonidae	Ichneumons	2		7
	Sphecidae	Sphecid wasps			3
	Vespidae	Vespid wasps	1		1
Orthoptera	Acrididae	Short-horned grasshoppers			13
	Gryllidae	Crickets			5
Plecoptera	Unknown	Stoneflies		1	2
Psocoptera	Liposcelidae	Liposcelid booklice		2	4
Thysanoptera	Thripidae	Thrips	21		36
Trichoptera	Hydroptilidae	Caddisflies			2

Post-burn Resprouting in Bigtooth Maple (*Acer grandidentatum*): A Case Study

Beth L. Corbin¹ and Douglas H. Page²

ABSTRACT

Bigtooth maple (Acer grandidentatum) is a common component of mountain shrub communities throughout Utah and is sometimes an abundant component in northern Utah. It typically grows with Gambel oak (Quercus gambelii), either as a co-dominant, or sometimes replacing Gambel oak on mesic sites. Historically, the oak/maple cover type is believed to have had relatively short fire-free intervals (perhaps no longer than 30-80 years and sometimes shorter). Today it is often the dominant vegetation type in wildland urban interface zones of northern and central Utah. There has been some perception that bigtooth maple is a poor sprouter following fire. However, we have observed prolific post-fire sprouting for bigtooth maple in northern Utah. This case study was conducted to quantify our observations. Study sites were associated with two prescribed fires and one wildfire in the Wasatch Mountains of Wasatch County, Utah. Pre and post-burn bigtooth maple sprouts were counted in 10 to 19, 0.004-ha (0.01-acre) plots per site. Post-burn sprout numbers and survival on all sites support the conclusion that bigtooth maple recovers from fire by vegetative regeneration from root and crown sprouting and is thus well adapted to relatively high frequency fire regimes. These findings should facilitate management of landscapes where bigtooth maple is a significant component.

INTRODUCTION

Bigtooth maple (*Acer grandidentatum*), also called canyon maple, is a common and sometimes abundant component within the mountain shrub community commonly called “oakbrush” (or oak/maple brush) in the forests and woodlands of Utah. It often grows with Gambel oak (*Quercus gambelii*), either as a co-dominant, or in its place, replacing Gambel oak in canyon bottoms and moister areas. In the northern Wasatch Mountains (Cache County, UT), Gambel oak is replaced entirely by bigtooth maple, which forms extensive stands. The oak/maple brush community is dominant along the Wasatch Front in the wildland urban interface zone, and is common upward into the Wasatch Mountains to about 2,450 m (8000 ft) in elevation.

Historically, the oak/maple cover type is believed to have had fire return intervals of about 40-60 years of primarily stand-replacing fires (Havlina 2003). Fire regimes in the oak/maple zone have been altered since settlement times by elimination of Native American burning, fire suppression,

and grazing removal of fine fuels (Bradley et al 1991). As a result, oak/maple stands may be more extensive than they were 75 to 150 years ago (Ibid).

Bigtooth maple is usually a multi-stemmed small tree or large shrub in Utah, typically 4-8 m (12-25 ft) tall but occasionally exceeding 12 m (40 ft) in height (Welsh et al 2003). It is deciduous, and often provides spectacular fall colors. The root system is shallow but wide-spreading. Although it produces abundant seed on average every 2 to 3 years, most reproduction is vegetative, by layering (Uchytel 1990). In the field, seedlings seldom survive (Tollefson 2006). Although all native maples will sprout from the base, the degree varies with species and plant age (Zasada and Strong 2003).

PROBLEM STATEMENT

The Forest Service’s Fire Effects Information System (www.fs.fed.us/database/feis) is a web-based database of taxonomy, distribution, management considerations, botanical and ecological characteristics, fire ecology, and fire effects references, by species. The Fire Effects Information System’s (FEIS) previous record for *Acer grandidentatum* was updated in 1990, and included very little information on post-fire sprouting (Uchytel 1990). Specifically, it suggested that bigtooth maple is a poor sprouter.

We have observed bigtooth maple sprouting prolifically after wildfires and prescribed burns on the Uinta and Wasatch-Cache National Forests. Our personal observations lead us to believe that the sprouting vigor of bigtooth maple is relatively close to that of Gambel oak, as the post-fire species mix appears to closely mimic the pre-fire mix. Thus we identified a need to document this response in order to provide reliable information to update the FEIS, and in 2004 collected the field data presented below.

The FEIS record has been recently updated, with a substantial increase in references, and includes good information on bigtooth maple’s sprouting ability (Tollefson 2006). Our observations (presented here) provide additional documentation on bigtooth maple’s sprouting response following fire.

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Beth Corbin, Uinta-Wasatch-Cache National Forest, Ogden, UT

²Douglas Page, Bureau of Land Management, Cedar City, UT

LITERATURE REVIEW

We conducted a literature review in 2004 to determine whether relevant published information exists that was either developed after the 1990 FEIS record, or was missed by the FEIS compiler. We found very little on bigtooth maple at all (for example: Buchmann et al 1997; Tankersley pers. Com¹), and nothing on its resprouting ability.

Work by Harper et al (1985) in central and northern Utah suggested that bigtooth maple may be invading Gambel oak stands on drier upland sites, presumably due to fire suppression. Harper et al state that:

Unfortunately, there appear to be no observations on either the relative frequency of fire along drainage ways and on adjacent open slopes in our area nor on the relative frequency and vigor of maple sprouting after fire on such contrasting sites (p 15).

By contrast, the 2006 FEIS record located several references on bigtooth maple sprouting following fire, including “vigorously” sprouting and stands that “return quickly” to prefire species composition (Tollefson 2006).

METHODS

We are documenting a case study of bigtooth maple sprouting following two prescribed fires and one wildfire in the Wasatch Mountains of the Uinta National Forest. The Cascade Springs I Prescribed Fire of April 2000 burned about 376 ha (930 acres) on National Forest lands. The Cascade II Prescribed Fire (and wildfire) of September 2003 burned about 3,150 ha (7,800 acres), mostly National Forest and State Park lands. The Wasatch Mountain wildfire of August 1990 burned about 1,200 ha (3,000 acres), mostly of Wasatch State Park lands. All fires are in Wasatch County, Utah, and within the Provo River watershed.

Sites were selected from each fire based on access, vegetation type (dominated by oak/maple), and fire intensity (overstory stand replacement areas). Two to three transects were sampled in each fire in August 2004. Transects were on specified compass directions from each starting point. Distances were measured by pacing. At 1-chain (20 m) intervals, a 0.004 ha (100th a) circular plot was checked. If no bigtooth maple were present (either sprouts, dead stems, or mature stems), a “no” was recorded and we moved on to the next plot. For plots that had or still contained maple, the plot center was moved to the nearest maple clump and the following was recorded:

- Pre-fire stems of maple: For each stem – diameter at 15 cm (6 in.) stump height, status (dead, healthy, unhealthy), and burn intensity [unburned, light charring (<25 percent of bole diameter at any point), mod/heavy charring (25-95 percent charring), intense charring (>95 percent bole charring)].
- Post-fire sprouts: Number of live sprouts, number of dead sprouts, average diameters and heights of live and dead sprouts 15 cm (6 in.). Dead sprouts were indicative of maple that sprouted post-fire, but did not survive.
- Basic topographic site information (ridgetop, hillslope, or drainage bottom; slope and aspect).

A minimum of 10 plots containing maple from each fire were evaluated. Up to 19 plots containing maple per fire were recorded. Plots were not permanently marked.

Data analysis consisted of comparing pre-burn and post-burn maple stem density, using a paired Student’s T-test. Assumptions were made for independent sampling and normal distributions with equal variation of the population data, which appears to be the case.

OBSERVATION SUMMARY

Sprouting

Pre-burn and post-burn stem data are summarized for the three fires in table 1. We found that nearly all (95 percent) of the plots with pre-burn bigtooth maple had post-fire sprouts. Results were similar for the percentage of clumps (95 percent) and individual stems (96 percent) that had associated sprouts. The number of stems per plot of post-burn sprouts was significantly higher than the pre-burn stems per plot, using a paired Student’s T-test. We did not find a correlation between pre-burn stem size and post-burn sprouting performance, but our few non-sprouting stems constitute a very small sample. Based on these numbers, it appears that bigtooth maple is well adapted to sprouting after fire.

One apparent seedling (second-year, but only about 1.5 inches tall) was found on a plot in the Wasatch Mountain Fire. Otherwise, all regeneration was from sprouts. Most sprouts were stump sprouts originating within about 15 to 30 cm (6 to 12 in.) of the stump), but some root sprouts originating more than 30 cm (12 in.) from the stump were also found.

A small number of dead sprouts were observed, mostly in the older (Wasatch Mountain) fire. Natural competitive self-thinning through age/size/density-related mortality on the high number of sprouts present is to be expected.

¹Tankersley, Boyce. e-mail to Beth Corbin July 30, 2004 re: summary of his Master’s Thesis work on Growth and Propagation of *Acer grandidentatum* Nutt. from Texas A&M University, 1981.

Intensity

Nearly all of the plots were intensely burned, and all pre-burn stems in the plots were dead and appeared to have been killed immediately by the fire. Most stems were rated as intensely burned, with a few rated moderate/heavy charring. In the Wasatch Mountain Fire, about half of the plots had sprouts where no parent stem was seen, presumably because either the stem was totally consumed, was small and had decomposed, had been covered by litter or soil, or it had fallen and rolled away in the intervening 16 years since the fire. In the more recent Cascade fires, standing dead maple was visible at all sprout clumps.

Topography

See table 2 for the number and percent of plots containing maple that were found in various topographic positions for each fire. No correlation between topographic position and resprouting was apparent, since plots in all topographic positions had sprouts. The two plots with maple that did not resprout were on lower slopes in the Cascade II fire. Not all topographic positions were equally or randomly sampled, since we chose overstory stand replacement areas with a high maple component.

CONCLUSIONS AND IMPLICATIONS

Based upon the plot data collected on these three fires in Wasatch County, Utah, we conclude that bigtooth maple is well adapted to resprouting after fire. It appears that its sprouting vigor is similar to that of Gambel oak. Sprouting vigor may be related to site quality and pre-fire stem vigor. It is reasonable to surmise that the sprouting ability of both bigtooth maple and Gambel oak is related to their adaptation to ecosystems where fire has played a frequent role.

ACKNOWLEDGEMENTS

We would like to thank the following individuals who took the time to review and make critical comments on drafts of this document. Dr. James N. Long, Professor of Silviculture, Utah State University. Stanley G. Kitchen, Research Ecologist, USFS, Provo. Lauren B. Shapiro, Fire Ecologist, Uinta National Forest, Provo.

Table 1—Sprouting observation summary.

Fire	Number of plots with maple	Number of maple plots with sprouting	Number of maple clumps with sprouting	Number of individual stems with sprouting	Average number of stems/plot pre-burn	Average number of stems/plot post-burn ^a	Average diameter of pre-burn stems (inches)
Wasatch Mountain (August 1990)	19 out of 22 sampled (86%)	19 out of 19 sampled (100%)	90 out of 90 observed (100%)	86 out of 86 observed (100%)	4.5 (range 0 to 17)	43.3 (range 4 to 131)	2.8" (range 0.6 to 6.5")
Cascade I (April 2000)	14 out of 70 sampled (20%)	14 out of 14 sampled (100%)	24 out of 24 observed (100%)	75 out of 75 observed (100%)	5.4 (range 1 to 13)	30.6 (range 8 to 87)	2.9" (range 0.2 to 6.4")
Cascade II (September 2003)	10 out of 16 sampled (63%)	8 out of 10 sampled (80%)	19 out of 26 observed (73%)	44 out of 53 observed (83%)	5.3 (range 2 to 12)	30.6 (range 0 to 99)	1.6" (range 0.2 to 5.3) ^b

^aThe difference between pre-burn number of stems per plot and post-burn number of stems per plot is statistically significant at $p = 0.05$ for all three fires.

^bPre-burn stems that sprouted ($n = 44$) averaged 1.6" diameter (range 0.2-5.3"), while stems that did not sprout ($n = 9$) averaged 1.4" diameter (range 0.4-3.7"). These differences are not statistically significant at $p = 0.1$.

Table 2—Topography. Plots with maple were found in the following topographic positions:

Fire	ridgetop	upper slope	mid-slope	lower slope	drainage bottom
Wasatch Mountain	3/19 16%		16/19 84%		
Cascade I		3/14 21%	1/14 7%	9/14 64%	1/14 7%
Cascade II				6/10 60%	4/10 40%

REFERENCES

- Bradley, A.F., Noste, N.V., Fischer, W.C. 1991 Fire ecology of forests and woodlands in Utah. Gen. Tech. Rep. INT-287. Ogden, UT: USDA Forest Service, Intermountain Research Station. 128 p.
- Buchmann, N., Kao, W-Y., Ehleringer, J. 1997. Influence of stand structure on carbon-13 of vegetation, soils, and canopy air within deciduous and evergreen forests in Utah, United States. *Oecologia*; Vol. 110, no. 1 (March 1997): 109-119.
- Harper, K.T., Wagstaff, F.J., Kunzier, L.M. 1985. Biology and management of the Gambel oak vegetation type: a literature review. General Technical Report INT-179. Ogden, UT: USDA Forest Service, Intermountain Forest and Range Experiment Station. 31 p.
- Havlina, D. 2003. Fire Regime Condition Class (FRCC) Interagency Handbook Reference Conditions for Interior chaparral (CHAP5). 8/12/03. http://www.frcc.gov/docs/PNVG/West/CHAP5_Description.pdf
- Tankersley, B. e-mail to Beth Corbin July 30, 2004 re: summary of his Master's Thesis work on Growth and Propagation of *Acer grandidentatum* Nutt. from Texas A&M University, 1981.
- Tollefson, J.E. 2006. *Acer grandidentatum*. In: Fire Effects Information System, [Online]. USDA, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). Available: <http://www.fs.fed.us/database/feis/> [accessed 2007, September 6].
- Uchytel, R.J. 1990. *Acer grandidentatum*. In: Fire Effects Information System, [Online]. USDA, Forest Service, Rocky Mountain Research Station, Fire Science Laboratory (Producer). Available: <http://www.fs.fed.us/database/feis/> [accessed 2004, August 10].
- Welsh, S.L., Atwood, N.D., Goodrich, S., Higgins, L.C. 2003. A Utah Flora, Third Edition. Provo, UT: Brigham Young University. 912 p.
- Zasada, J.C., Strong, T.F. 2003. *Acer L.* In: Bonner, F.T. and Nisley, R.T. Woody Plants Seed Manual. Forest Service, Washington, DC. 24 p.

Big Sagebrush in Pinyon-juniper Woodlands: Using Forest Inventory and Analysis Data as a Management Tool for Quantifying and Monitoring Mule Deer Habitat

Chris Witt¹ and Paul L. Patterson¹

ABSTRACT

*We used Interior West Forest Inventory and Analysis (IW-FIA) data to identify conditions where pinyon-juniper woodlands provide security cover, thermal cover, and suitable amounts of big sagebrush (*Artemisia tridentata* spp.) forage to mule deer in Utah. Roughly one quarter of Utah's pinyon-juniper woodlands had a big sagebrush component in their understory. Security cover was the least abundant habitat component investigated and only an estimated three percent of Utah's pinyon-juniper contained forage, security, and thermal cover concurrently. Area of potential mule deer habitat was generally distributed within ecoregion provinces in Utah in proportion to the spatial extent of each province. Quantile regression analysis suggests that when there is less than 22 percent crown cover, more than 20 percent of pinyon-juniper woodlands can be expected to have greater than 15 percent sagebrush cover. We demonstrate the utility of FIA data as an estimator of current habitat features and a long-term monitoring tool and show how quantile regression can be a useful tool for analyzing data with heterogeneous variation.*

INTRODUCTION

Adaptive Management of ecological systems at the regional scale has recently become popular with resource managers in the United States (Johnson and others 1999). These efforts have had varying degrees of success and have always been expensive to implement. Many of the failures of these plans can be attributed to lack of scientific data, inability to interpret data meaningfully, and inconsistencies in data analysis (Arnett and Sallabanks 1998). The funding limitations and small temporal and spatial scale at which most research is conducted also restricts the applicability of these efforts to larger comprehensive projects. As such, it is not surprising that difficulties persist in the collection and application of knowledge to create management tools that are flexible and can be applied at large spatial and temporal scales.

The Forest Inventory and Analysis (FIA) program of the Forest Service, U.S. Department of Agriculture, is responsible for assessing the status and trends of all forested lands in the U.S. (Gillespie 1999). The FIA program conducts inventories on all forested lands of the U.S. using a nationally standardized plot design (figure

1) at an intensity of approximately one field plot per 2,388 ha (5,800 ac). The Intermountain West region of FIA (IW-FIA) is responsible for collecting data on plots in Arizona, Colorado, Idaho, Montana, Nevada, New Mexico, Utah, and Wyoming. In the states where FIA is fully funded, field crews visit roughly ten percent of a state's plots every year and measure up to 140 different variables on each plot. Plots belonging to each annual panel are distributed across each state so as to be free of geographic bias. The measurements are used to calculate estimates of area, age distribution, growth, mortality, volume, and biomass among others. Once a full cycle is completed, another begins and trend data can be analyzed. The extent, consistency, and perpetual nature of FIA data lends itself to quantifying resources over large areas and monitoring changes in those resources over time.

FIA information can be useful in quantifying habitat components for species that utilize forested environments at some point in their life cycle. One such species is the mule deer (*Odocoileus hemionus*). Mule deer are an important wildlife species in the Intermountain West and recent evidence has suggested population declines in several states (Stewart 1999; Unsworth and others 1999; Wasley 2004). Several reasons for population declines have been suggested including drought, harsh winters, predation, change in vegetative communities and overgrazing by domestic livestock (Wasley 2004; Gill and others 2001; Pojar and Bowden 2004). Mule deer occupy many environments and utilize different habitats as season and resource availability permit. Across much of their range, mule deer spend the warm growing season at higher elevations, taking advantage of seasonally abundant herbaceous vegetation, then gradually migrate to lower elevation shrublands as temperatures cool, food resources disappear, and snow accumulates. The time spent in the transitional areas between summer and winter range can be very important to mule deer, especially where winter or summer ranges have been degraded, fragmented or otherwise isolated (Short 1981; Verme and Ullrey 1984; Sawyer and others 2005). In the Intermountain West, much of this transitional range consists of pinyon-juniper woodlands. The plant community structure in these woodlands can vary dramatically but where certain conditions are met, they can provide important habitat to mule deer. The structure provided by woody shrubs and small trees in these woodlands can be important in providing evaluating security and thermal cover for mule

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Chris Witt, Paul L. Patterson, Interior West Forest Inventory and Analysis; Rocky Mountain Research Station, USFS

deer. Big sagebrush species (*Artemisia tridentata* spp.) are sometimes found in pinyon-juniper woodlands and are known to be an important food resource for mule deer (Reynolds 1960; Papez 1976; Wambolt and others 1987; Wambolt 1996; Welsh 2005). The relationship of pinyon and juniper tree cover and the extent of big sagebrush and other woody vegetation has been explored for western juniper woodlands (Miller and others 2000; Miller 2001) but identifying general relationships in pinyon-juniper woodlands using a landscape-level dataset has not been addressed to a great extent.

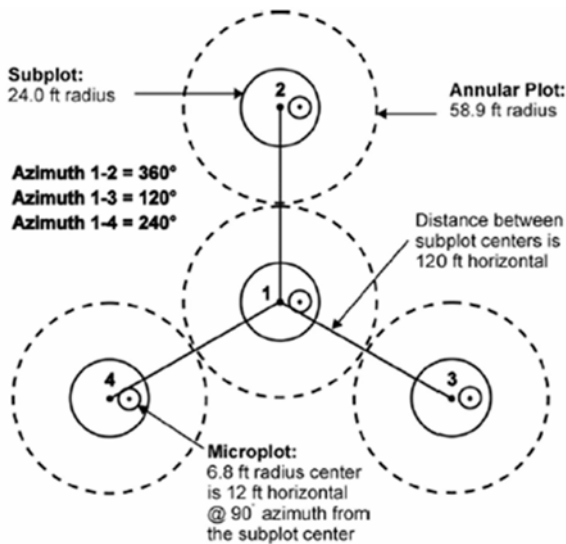


Figure 1—Standard Forest Inventory and Analysis Phase 2 plot diagram.

In this study we demonstrate the utility of FIA data as a census and monitoring tool when applied to mule deer habitat over large areas. We use Interior West Forest Inventory and Analysis data for Utah from 2000 to 2007 to estimate has of pinyon-juniper woodlands that provide security cover, thermal cover, and forage in the form of big sagebrush (*Artemisia tridentata* spp.) across the state and by the five major ecoregion provinces found in Utah as described by Bailey (1995). Since FIA plots are uniformly distributed geographically, we can translate statements involving percentage of plots into percentage of land. Area estimates are produced using methods outlined in Bechtold and Patterson (2005). We also use quantile regression (Cade and Noon 2003; Koenker 2005) to identify thresholds of tree crown cover as it relates to potential cover of sagebrush available to mule deer.

DATA AND METHODS

The FIA program conducts inventory on all forested lands of the U.S. at an intensity of approximately one field plot per 2,308 ha. FIA defines “Forest” as those lands 0.4 ha or greater in area and at least 36.6 m wide having five percent or greater crown cover of tree species tallied by FIA crews

(USDA 2007). Where plots meet these criteria, IW-FIA also estimates aerial cover of understory vegetation by dominant species. Dominance is defined as those occurring at or greater than five percent cover on any subplot. Field crews also estimate cover by major life form (graminoid, forb, shrub and trees 12.7 cm or smaller diameter at breast height [DBH]) within three different horizontal layers to provide additional information on forest structure.

Crown cover on plots is estimated using four 7.62 m transects running in the cardinal directions from each subplot center. The presence or absence of aerial cover of tally trees 2.54 cm and greater diameter at breast height (DBH) is recorded every 30.48 cm along each transect. Percent cover of tree and shrub understory from two layer classes (47 to 183 cm and > 183 cm) is estimated on each 0.017 ha subplot using techniques described in USDA 2007. Cover of individual species is recorded if estimated aerial cover meets or exceeds five percent of the subplot area in a particular condition (in this case a pinyon-juniper forest type). We defined big sagebrush as being present if it occurred with at least five percent cover across the plot (or portion of the plot in the pinyon-juniper condition) while those having less than five percent cover of sagebrush were classified as having none.

Based on research by Armleder and others (1994), Tull and others (2001), and Miller (2001), we considered adequate thermal cover to be equal or greater than 35 percent on the plot. We used either crown cover or the combined tree and shrub cover in layer 3 (whichever was greater) to quantify the minimum thermal cover present on a plot. Security cover was defined as vegetative cover greater than 47 cm tall. We calculated security cover on a plot by combining understory cover of trees and shrubs in layer 2. We considered stands with 40 percent or greater crown cover to provide adequate security cover given the low crown morphology and multiple stemmed growth form often exhibited by individual pinyon and juniper trees (Miller 2001; Reading 2004). The amount of big sagebrush that mule deer need in winter depends on many factors including extent of cover, condition of the plants, height in relation to typical snow depth, size of deer herd, and the presence and abundance of additional food resources. In this study, we used plots with at least five percent sagebrush cover to calculate area estimates and used 15 percent sagebrush cover for the purposes of discussing the applications of quantile regression.

For a threshold of big sagebrush cover, we explore the issue of determining a cutoff value of percent crown cover in pinyon-juniper woodlands, where for large percentage of land with crown cover greater than the cutoff value we have a high level of confidence that the percentage of big sagebrush cover is below the user determined threshold. For example, if the user determined threshold is 15 percent, then we would like to make statements of the form, “In

pinon-juniper woodlands that have a big sagebrush component in their understory and crown cover less than “X” percent, we are 95percent confident that “Y” percent of this land has more than 15 percent big sagebrush cover.” Since we are interested the tails of the probability distribution of big sagebrush cover, the standard technique of finding the mean of the conditional distribution of big sagebrush cover as a linear function of the crown cover in pinon-juniper woodlands is of limited usefulness; instead we used quantile regression.

Quantile regression is a technique for estimating the quantiles of the conditional distribution of a response variable as function of a set of predictor variables. Although quantile regression methods have been investigated since the early 18th century, implementation for nonparametric error distributions only occurred with advent of modern computing (Koenker 2005). Quantiles can be viewed as a decimal expression of percentiles. For example, the 80th percentile would be expressed as the 0.8 quantile. For a primer on the uses of quantile regression for ecological processes see Cade and Noon (2003). For this analysis we assumed the heteroscedastic linear model:

$$y_i = \beta_0 + \beta_1 x_i + \sigma(x_i) \varepsilon_i$$

where y is the response variable, X is the predictor variable, β_0 and β_1 are fixed parameters and the $\{\varepsilon_i\}$ are independent identical distributed random errors. Quantile regression has the advantage of accommodating heterogeneous variation in the conditional distributions and not requiring parametric error distributions. We used the **quantreg** package in version 2.5.1 of the R language to conduct the quantile regression analysis.

RESULTS

Area Estimates

From 2000 to 2007, FIA collected data on 1,618 conditions (plots or portions of plots) of pinon-juniper woodland in Utah. Of these, 424 had big sagebrush cover estimated at or greater than five percent. Because we removed 3 plots from analysis because of obvious errors in crown cover estimates, area estimates and quantile regression are based on 421 plots. Of the estimated 4,568,265 ha of pinon-juniper woodland in Utah, 1,192,778 ha (26 percent) have a big sagebrush understory component (figure 2). Roughly 212,622 ha (18 percent) of the area estimated to have big sagebrush met the requisites for thermal cover while and estimated 142,495 ha (12 percent) met the requirements for adequate security cover. There are roughly 137,120 ha in Utah that simultaneously provide security cover, thermal cover and big sagebrush forage at or greater than five percent. This represents three percent of Utah’s pinon-juniper woodlands and 12 percent of pinon-juniper woodlands having big sagebrush in them.

The portion of the *Intermountain Semidesert and Desert Province* (341) in Utah is estimated to have the greatest total area of pinon-juniper with big sagebrush (560,064 ha). This Ecoregion provides the largest estimated area having security cover (62,045 ha) and thermal cover (91,440 ha) as well (figure 3). The *Intermountain Semidesert Province* (342) ha the largest percentage of its pinon-juniper woodlands having a big sagebrush component (33 percent). In addition, M341 ha the greatest percentage of its pinon-juniper woodland providing thermal (26 percent) and security cover (16 percent) in Utah. The pinon-juniper woodlands that provide big sagebrush, thermal cover, and security cover concurrently also reside largely in these two ecoregions.

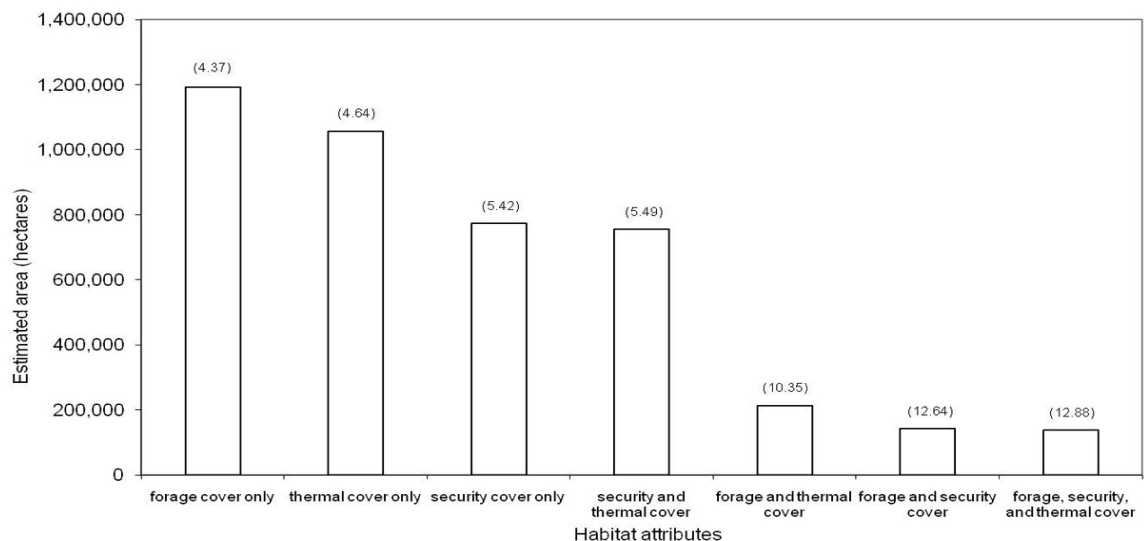


Figure 2—Estimated number of hectares meeting forage, thermal cover, security cover and various combinations of these habitat components in pinon-juniper woodlands in Utah. Numbers in () reflect percent standard error.

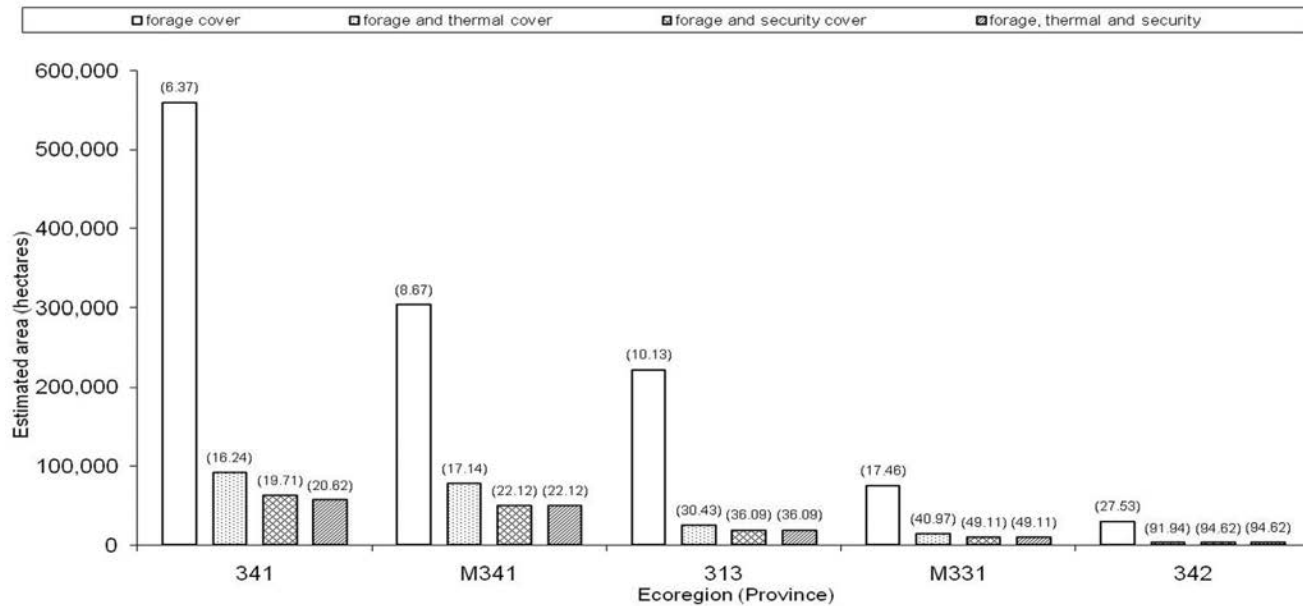


Figure 3—Estimated number of hectares of forage (*Artemisia tridentata* spp.) and a combination of forage and other habitat components in pinyon-juniper woodlands by major ecoregion provinces in Utah. Numbers in () reflect percent standard error.

Cover Thresholds

There were 421 plots used in the quantile regression analysis of big sagebrush cover as a function of crown cover in pinyon-juniper woodlands. A plot of big sagebrush cover versus crown cover indicates there may be a decrease in big sagebrush cover as crown cover increases (figure 4). The graph also contains the quantile regression lines for 90th, 80th, 70th and 50th percentile and the least squares estimate of the mean.

The upper and lower 95percent confidence bounds for the 0.8 quantile of the conditional distribution calculated at selected integer values of the percent crown cover (figure 5). For the 0.9 quantile only the lower confidence bound was calculated at the same selected integer values.

DISCUSSION

Area Estimates

Pinyon-juniper woodlands that provide adequate thermal and security cover as well a big sagebrush component for mule deer appear to be uncommon in Utah. Only three percent of Utah's pinyon-juniper stands meet others of the forage, thermal cover, and security cover thresholds outlined in this study. The understory structure thought to be necessary to provide mule deer cover from detection by predators appears to be the least common attribute analyzed here, evident in only 12 percent of the stands that had a big sagebrush component. However, 17 percent of all pinyon-juniper woodland in Utah had adequate security cover. This suggests that shrub and tree species other than big sagebrush are contributing to security cover in many pinyon-juniper stands. There may also be other food resources in these stands as several species other than

sagebrush provide forage for mule deer. Therefore, we can not say that the pinyon-juniper stands that have no sagebrush are not providing a food resource to mule deer. It is probable that some portion of the land area having good security cover but no sagebrush is providing food in another form.

Pinyon-juniper stands providing thermal cover were almost as common as those having a big sagebrush component (1,057,031 ha and 1,192,778 ha respectively). However, only 18 percent of pinyon-juniper stands shared both qualities. This was not unexpected as it is known that big sagebrush does not perform well in shaded areas and it is shade from the tree and tall shrub canopy that provides thermal insulation to mule deer during times of temperature extremes. Most big sagebrush does not grow to a height where it would contribute to thermal cover (greater than or equal 183 cm) so any tree or shrub providing thermal cover is likely in direct competition with big sagebrush for important resources.

The standard area estimates produced by FIA data can identify limiting resources for species that utilize resources at large spatial scales. In this case, we have identified mule deer security cover as being less abundant than thermal cover in pinyon-juniper/big sagebrush woodlands. If winter stress on mule deer has been identified as a larger concern than predation, a resource manager may wish to apply management strategies that promote understory growth and limits crown closure. FIA data can then be used to monitor the overall effects on the landscape through time to help maintain a balance between the multiple needs of mule deer populations as well as habitat for other species.

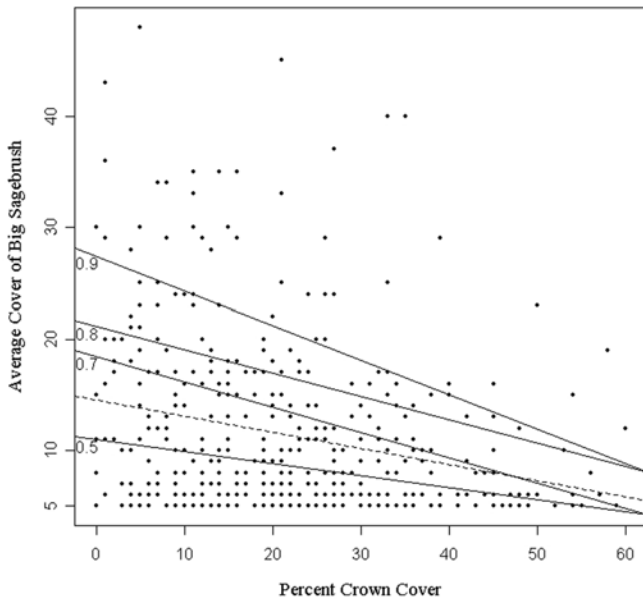


Figure 4—Average percent cover of big sagebrush (y) versus percent crown cover in pinyon-juniper woodlands (X) on $n = 421$ plots, the 0.9, 0.8, 0.7, and 0.5 regression quantile estimates (solid lines), and the least squares estimate of mean (dashed line).

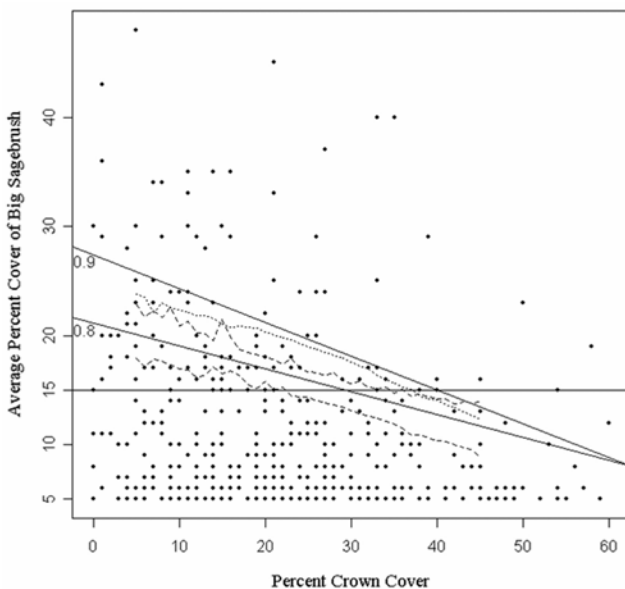


Figure 5—Average percent cover of big sagebrush (y) versus percent crown cover in pinyon-juniper woodlands (X) on $n = 421$ plots, the 0.8 and 0.9 regression quantile estimates and $y = 15$ line (solid lines). The dashed lines connect the upper and lower 95% confidence bounds of the 0.8 quantile at integer values of percent crown cover. The dotted line connects lower 95% confidence bounds of the 0.8 quantile at integer values of percent crown cover.

Although beyond the scope of this study, it would be interesting to see how the FIA plots with big sagebrush on them were juxtaposed with non-forested sagebrush communities. If these plots occur along ecotones more often than in insular stands, it could have an influence on management of both communities. Pinyon-juniper expansion into shrublands is a much debated issue in the West, often because of the impacts on sagebrush obligates (Blackburn and Tueller 1970; Burkhardt and Tisdale 1976; Soule and Knapp 1999). Management prescriptions for invading pinyon and juniper often include fire and mechanized thinning or removal along the woodland-shrubland interface in order to retard, halt, or reverse the leading edge of woodland expansion. Given the general process of invasion, establishment and succession of vegetative communities, it is likely that these edges include much of the woodland area that has a sagebrush component and other attributes beneficial to mule deer. Managers may be improving habitat for one species of concern while unknowingly degrading the habitat of another.

As a monitoring tool, remeasurement of a full cycle of plots provides the best information on large scale trends in forest systems. However, a full 10 year cycle of FIA data is not necessary to detect profound changes in forest structure. Some drought-related responses of pinyon pine (Shaw 2005; Shaw and others 2005) and other conifers (Thompson, unpublished data) have been detected by exploring several sequential years of FIA data. This allows land managers to monitor responses to management prescriptions and disturbances at shorter time intervals. They can then modify their approach should undesirable trends become evident. This is a key feature of adaptive management and it is only possible when baseline information and consistent post-implementation data are readily available.

Quantile Regression

The upper and lower confidence bounds for conditional quantiles can be used to inform management decisions. For example suppose it is desirable to have more than 15 percent cover of big sagebrush. For the estimated 0.8 quantile, an examination of the lower 95percent confidence bounds at integer values of crown cover indicate that for crown cover less than 22 percent the lower 95percent confidence bound is greater than 15 percent (figure 5). So we are 95percent confident that 20 percent of the pinyon-juniper lands with a presence of big sagebrush and with crown cover less than 22 percent have more than 15 percent crown cover of big sagebrush. Based on the estimated 0.9 quantile lower 95percent confidence bounds, we are 95percent confident 10 percent of the pinyon-juniper lands with a presence of big sagebrush and with crown cover less than 38 percent have more than 15 percent crown cover of big sagebrush. The 95percent confidence bounds are not simultaneous confidence bounds, so overall confidence maybe less than 95percent.

An examination of the upper 95percent confidence bounds for the 80th percentile at integer values of crown covers indicate that for crown cover greater than 37 percent the upper 95percent confidence bound is less than 15 percent (figure 5). In other words we are 95 percent confident that 80 percent of the pinyon-juniper lands having a big sagebrush component and with crown cover greater than 37 percent have less than 15 percent cover of big sagebrush.

Since thermal cover is often provided only by tree crown cover in pinyon-juniper woodlands (as opposed to sub-crown vegetation), we can use the quantile regression analysis to say something about the frequency at which we can expect to see both sagebrush forage and thermal cover provided concurrently. Figure 5 illustrates that at 40 percent crown cover, the threshold at which thermal cover is assumed to be provided, one could expect 80 percent of these woodlands to have less than about 14 percent sagebrush cover and 20 percent of them to have roughly 10.5 percent or greater sagebrush cover (the confidence level of this statement would be 90 percent). Whether or not this is enough area for mule deer depends on a multitude of other factors, but at least the resource manager is armed with reasonable expectations regarding the amount of crown cover and associated sagebrush cover potentially available in pinyon-juniper woodlands.

As with any regression analysis there are pitfalls in using the estimates outside the range of the data. In the case the of heterogeneous variance model, the slopes of the regression line can be different and at some point the lines will intersect. Figure 4 indicates this happens for the quantile regression lines with equal 0.80 and 0.90, when the crown cover is slightly greater than 60 (in other words,, for crown cover slightly greater than 60) the estimated quantiles have the disconcerting property that the 80th percentile is greater than the 90th percentile. This is a possible pitfall when pushing quantile regression estimates beyond the range of the data.

CONCLUSION

FIA data can produce area estimates for habitat features for regional or landscape analyses. Habitats of animal species that utilize resources at these scales can be monitored over time by comparing past estimates of habitat feature to present ones. In this case, wildlife managers can use FIA data to monitor the amount and extent of pinyon-juniper woodlands that provide one or many structural features used by mule deer in Utah. However, these types of analyses can be performed for any habitat feature in any forest type measured by FIA field crews. FIA currently collects data on those lands that meet the criteria for forested land, but there are several ongoing initiatives and pilot studies exploring the utility of FIA protocols on rangelands and urban landscapes. It is possible that at some

point in the future, FIA data can be applied to the spatial and temporal questions of habitats on all lands in the United States.

Oftentimes the mean of a distribution may not identify interesting relationships that can be useful to managers. Quantile regression is a useful tool for ecologists in that it allows exploration of all portions of a distribution in the presence of heterogeneous variance common in large-scale vegetation data. Looking at meaningful quantiles can help identify relationships and develop predictive models useful to resource managers and others interested in landscape ecology.

REFERENCES

- Armleder, H.M., Waterhouse, M., Keisker, D., Dawson, R. 1994. Winter habitat use by mule deer in the central interior of British Columbia. *Canadian Journal of Zoology*. 72: 1721-1725.
- Arnett, E.B., Sallabanks, R. 1998. Land manager perceptions of avian research and information needs: A case study. Pages 399-413 in J. M. Marzluff and R. Sallabanks editors. *Avian conservation*. Island Press. Washington, D.C. USA.
- Bailey, R.G. 1995. Description of the ecoregions of the United States (2nd ed) Misc Pub No.1391 Map scale 1:7,500,000. Forest Service. 108 p.
- Bechtold, W.A., Patterson, P.L. 2005. The enhanced Forest Inventory and Analysis program-national sampling design and estimation procedures. Gen. Tech. Rep. SRS-80. Ashville, NC: USDA, Forest Service, Southern Research Station. 85 p.
- Blackburn, W.H., Tueller, P.T. 1970. Pinyon and juniper invasion in black sagebrush communities in East-Central Nevada. *Ecology*. 51(5): 841-848.
- Burkhardt, W.J., Tisdale, E.W. 1976. Causes of juniper invasion in Southwestern Idaho. *Ecology*. 57(3): 472-484.
- Cade, B.S., Noon, B.R., 2003. A gentle introduction to quantile regression for ecologists. *Frontiers in Ecology and the Environment*. 1: 412-420.
- Gill, R.B. 2001. Declining mule deer populations in Colorado: reasons and responses. Colorado Division of Wildlife Special Report 77.
- Gillespie, A. Jr. 1999. Rationale for a national annual inventory program. *Journal of Forestry*. 97: 16-20.
- Johnson, B.L. 1999. The role of adaptive management as an operational approach for resource management agencies. *Conservation Ecology* 3(2): 8 [online] URL:<http://www.consecol.org/vol3/iss2/art8/>.
- Koenker, R. 2005. Quantile regression. New York: Cambridge University Press. 349 p.
- Miller, R.F., Svejcar, T.J., Rose, J.A. 2000. Impacts of western juniper on plant community composition and structure. *Journal of Range Management*. 53(6): 574-585.
- Miller, R.F. 2001. Woodland fish and wildlife: Managing western juniper for wildlife. MISCO286. Washington State University Cooperative Extension. 8 p.
- Papez, N.J. 1976. The Ruby-Butte deer herd. *Biological Bulletin* 5. Reno. Department of Fish and Game. 60 p.
- Pojar, T.M., Bowden, D.C. 2004. Neonatal mule deer fawn survival in West-Central Colorado. *Journal of Wildlife Management* 68(3): 550-560.
- Reading, H.A. 2004. Landscape assessment for pinyon-juniper woodlands within the North Kaibab Ranger District with special emphasis on wildlife habitat. Version 1.0. 106 p.
- Reynolds Jr. T.A. 1960. The mule deer: Its history, life history and management in Utah. Utah Department of Fish and Game. Publication No. 60-4.
- Sawyer, H., Lindzey, F., McWhirter, D. 2005. Mule deer and pronghorn migration in western Wyoming. *Wildlife Society Bulletin* 33(4): 1266-1273.
- Shaw, J.D., Steed, B.E., Deblander, L.T. 2005. Forest Inventory and Analysis (FIA) answers the question: What is happening to pinyon-juniper woodlands? *Journal of Forestry*. 103(6): 280-285.

- Shaw, J.D. 2005. Drought-related mortality in pinyon-juniper woodlands: A test case for the FIA annual inventory system. 2004 Proceedings of a symposium, 2004 September 21-24, Denver CO. Gen. Tech. Report WO-70. Washington D.C.: USDA, Forest Service. 126 p.
- Short, H.L. 1981. Nutrition and metabolism. Pages 99-127 *in* O.C. Wallmo editor. Mule and black-tailed deer of North America. University of Nebraska Press, Lincoln, NE, USA.
- Stewart, R. 1999. Mule deer. Utah Division of Wildlife Resources. Wildlife notebook series No. 13.
- Soule, P.T., Knapp, P.A. 1999. Western juniper expansion on adjacent disturbed and near-relict sites. *Journal of Range Management*. 52(5): 525-533.
- Tull, J.C., Krausman, P.R., Steidl, R.J. 2001. Bed-site selection by desert mule deer in southern Arizona. *The Southwest Naturalist*. 46(3): 354-357.
- Unsworth, J.W., Pac, D.F., White, G.C., Bartman, R.M. 1999. Mule deer survival in Colorado, Idaho, and Montana. *Journal of Wildlife Management*. 63(1):315-326.
- USDA 2007. Interior West Forest Inventory and Analysis field procedures. Version 3.01. USDA, Forest Service, Rocky Mountain Research Station. 420 p.
- Verme, L.J., Ullrey, D.E. 1984. Physiology and nutrition. Pages 91-118 *in* L. K. Hall, editor. White-tailed deer: ecology and management. Stackpole Books, Harrisburg, PA, USA
- Wambolt, C.L., McNeal, A.F. 1987. Selection of winter foraging sites by elk and mule deer. *Journal of Environmental Management*. 25: 285-291.
- Wambolt, C.L. 1996. Mule deer and elk foraging preferences of four sagebrush taxa. *Journal of Range Management*. 49: 499-503.
- Wasley, T. 2004. Nevada's mule deer. Population dynamics: issues and influences. *Biological Bulletin* No. 14.
- Welsh, B.L. 2005. Big sagebrush: A sea fragmented into lakes, ponds, and puddles. Gen Tech Rep. RMRS-144. USDA, Forest Service, Rocky Mountain Research Station. 210 p.

A Landscape Level Habitat Survey of Mule Deer Winter Range in Eastern Montana

William H. Thompson¹, Paul L. Hansen¹, and Michael R. Frisina²

ABSTRACT

*In 2006 and 2007 Environmental Solutions Group, LLC (ESG) conducted landscape level surveys of mule deer (*Odocoileus hemionus*) winter range to test new field methods for assessing degree of browse utilization, browse species condition, and large animal distribution patterns. In 2007, a site was surveyed on Big Sheep Mountain in the Cherry Creek mule deer census area north of Terry in eastern Montana, covering approximately 6,400 ac (2,590 ha). Methodology followed Frisina and Knapp (2006). Data were collected along foot travel routes in a quarter mile pattern to cover the study area, totaling approximately 54 mi (86 km) of route at each site. Each data point represents a route segment up to 0.25 mi (0.4 km) long having a single type of vegetation. Data were collected for canopy cover and utilization level by individual browse species, canopy cover of other major individual plant species, amount of bare ground, and frequency of fecal pellet group observed for each ungulate species. Potential users of this methodology may now assess whether it offers useful and cost-effective application for their purposes.*

INTRODUCTION

Aerial surveys of large ungulate populations on important winter ranges in Montana has long provided the Montana Department of Fish, Wildlife and Parks (MT FWP) with information on population dynamics for management purposes, but these data would be more useful if they could easily and rapidly be related to detailed habitat conditions at a landscape scale. A new method for rapidly assessing conditions of browse vegetation across large tracts of mule deer (*Odocoileus hemionus*) winter range habitat was proposed by MT FWP scientists (Frisina and Knapp 2006). There was need for a cost-effective way to document important habitat conditions, to detect trends, and to develop an on-the-ground gauge of ungulate utilization of the browse resource. This method employs rapid assessment techniques to cover a large area quickly, and this report documents the results of a test of the methodology. Professional managers may now examine the methodology and these results to determine whether it can provide sufficient useful information on habitat conditions to enable management decisions based on resource realities in a cost-effective way.

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹William H. Thompson and Paul L Hansen, Ecological Solutions Group, LLC, Stevensville, MT

²Michael R. Frisina, Range Coordinator, Montana Department of Fish, Wildlife and Parks, Butte, MT

METHODOLOGY

This methodology was designed as a rapid way to assess browse species composition, distribution, utilization, condition, and the distribution patterns of wild ungulates, as indicated by fecal pellet densities. The purpose of this project was to apply the technique so that MT FWP could assess the effectiveness of the proposed methodology. As a rapid assessment tool, ocular estimation techniques are employed. These techniques require trained field observers and generally follow ocular estimation methods described by US Forest Service (1993).

Parameters evaluated were:

- Canopy cover (abundance) of browse forage by species;
- Canopy cover (abundance) of dominant non-browse forage by species;
- Intensity of utilization of browse forage by species;
- Wild ungulate fecal pellet group frequency by species;
- Vegetation form (for example shrubland, grassland, forest, riparian, unvegetated)
- Amount of other ground cover (for example rock, bare ground, litter, etc.); and
- Landscape position (for example slope, benchland, broken topography, riparian bottom).

The result can be used to produce a series of maps of the study area showing:

- Distribution and pattern of utilization intensity of each browse species;
- Distribution of wild ungulate species usage of the site, based on fecal pellet group frequency; and
- Any of the various other site characteristics listed above.

Following Frisina and Knapp (2006), sampling routes were laid out on a map of the study area in a pattern to cover the area with a grid of approximately 0.25 mi (0.6 km) separation. The goal was to sample along at least 45 mi (72 km) of route on the study area. During the sampling process, the routes were broken into “station” intervals wherever vegetation form changed, or when the maximum station length of 0.25 mi (0.4 km) was reached. As the observer progressed along the route, station data records were completed before proceeding to the next station. A station data record represents observation of conditions within a narrow, linear, band 6 ft (2 m) wide, centered along the route path. Location coordinates were recorded at each station end.

Browse forage species and major non-browse forage species were listed while walking the route. Note was taken of all the other parameters of interest, and photos are taken along the route path to document conditions on the station. Upon reaching the station end, canopy cover estimates were entered for each listed browse and non-browse species, as well as for bare ground, amount of litter, etc. The observed level of browsing intensity was entered for each browse species, and the observed category of fecal pellet group frequency was entered for each wild ungulate species known to use the study area. The station was also characterized by categories of landform position and major vegetation form, and narrative commentary was entered to describe other observations. Table 1 further defines the data collected.

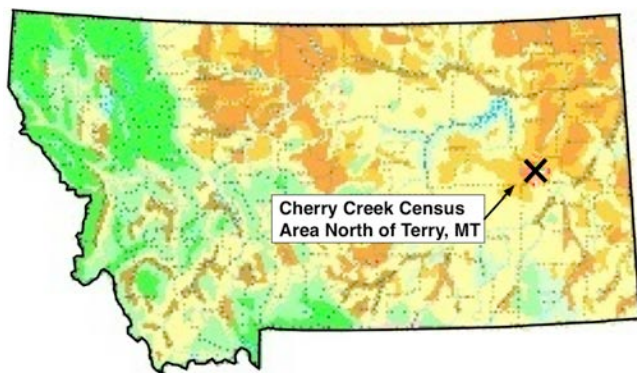


Figure 1—The Cherry Creek Mule Deer Census study area general location in Montana.

THE STUDY AREA

This study area in eastern Montana (figure 1) was chosen by MT FWP for trial of this methodology. The study area covers 10 sections (6,400 ac, or 2,591 ha) of prime mule deer winter range habitat. This paper describes the process and results applied to and produced on the study area surveyed in July 2007 on Big Sheep Mountain in the Cherry Creek Mule Deer Census Area north of Terry in eastern Montana (figure 2).

The study area is located in a highly varied terrain that includes wide ephemeral riparian bottoms, wooded draws with steep sides holding dense thickets of deciduous trees and shrubs, areas of erosional “badland” topography, and flat to moderately steep grassland steppe with scattered low shrubs. Elevation ranges from about 3,600 ft (1,312 m) at the west end down to about 3,100 ft (1,130 m) on the northern edge. The wooded draws and riparian bottoms support extensive stands of the Green Ash/Common Chokecherry (*Fraxinus pennsylvanica*/*Prunus virginiana*) habitat type (Hansen and others 1995). Vegetation in these communities is potentially very diverse, but historic and ongoing livestock use has severely altered most of the stands. Stands that have escaped being trampled and altered are physically protected by either steep terrain or a BLM livestock exclosure plot fence.

This site is in valuable winter range habitat for which MT FWP maintains mule deer (*Odocoileus hemionus*) population data. Pronghorn antelope (*Antilocapra americana*) and white-tailed deer (*Odocoileus virginianus*) also inhabit the area. Sharp-tailed grouse (*Tympanuchus phasianellus*) and sage grouse (*Centrocercus urophasianus*) use the shrub-grassland range, as well. Figure 3 shows the route layout as sampled at Big Sheep Mountain.

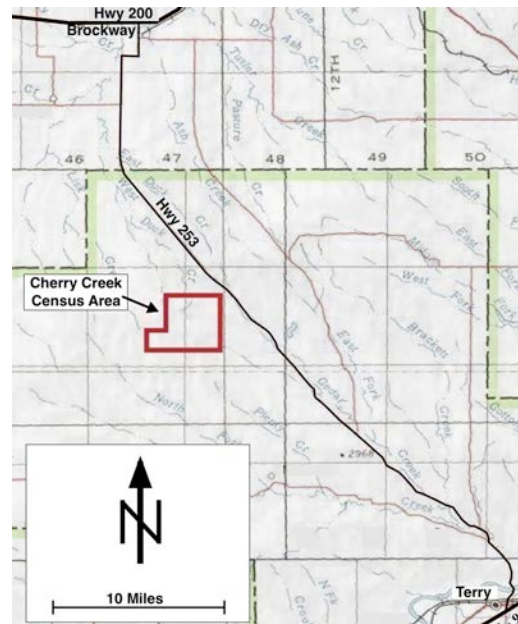


Figure 2—The vicinity of the Cherry Creek Mule Deer Census study area north of Terry in eastern Montana.

RESULTS

Browse Species Occurrence And Distribution

Of prime importance in this habitat survey is to obtain detailed knowledge of what species of browse are present on the study area, their abundance, condition, patterns of distribution, and the level to which they are being browsed. At each station, each browse species was recorded with estimated canopy cover on the station and with level of browsing intensity (Keigley and Frisina 1998). A list of browse species and data on their abundance in the study area is provided in table 2.

By weighting the species canopy cover station values by length of station, one may compile the abundance of each species over the study area at a spatial resolution determined by the density of routes and length of stations. This browse species information can be mapped by using the station location coordinates to give managers detailed tools by which to evaluate range conditions. Figure 4 shows the distribution of occurrence of fragrant sumac (*Rhus aromatica*) in the study area, as well as the distribution of browse intensity on the species.

Table 1—Variables sampled on each station polygon.

Variable Name	Variable Type	Unit of Measure	Note
Station Length	Numeric	Percent	Linear Measure
Browse Species Canopy Cover	Numeric Category ^a	Percent	Canopy Cover (Daubenmire 1959) ^b
Other Plant Species and Ground Cover	Numeric Category	Category	Canopy Cover (Daubenmire 1959)
Browse Species Level of Utilization	Attribute	Category	Intense, Light-to-moderate ^c
Vegetation Form	Attribute	Category	Shrubland, Grassland, Riparian ^d Forest
Landscape Position	Attribute	Category	Slope, Benchland, Broken Topography, Wooded Draw, Riparian ^e
Slope Steepness	Attribute	Category	Flat, Gentle, Moderate, Steep ^f
Ungulate Fecal Pellet Group Frequency	Attribute	Category	High, Moderate, Low, Limited ^g

^aThese categories represent a range of value, for which the range midpoint is used in calculations.

^bCanopy cover is the gross outline of an individual plant's foliage. Cover of each species is estimated as a fraction of the sampled area. The sum of all species cover may exceed 100 percent due to layering.

^cLevel of browse utilization on each species is in one of these two categories (Keigley and Frisina 1998)

^dVegetation form is based on the dominant tallest plant life form characterizing the station as: **Forest** = A station with at least 10 trees per acre; **Shrubland** = A station with at least 15 percent total canopy cover of shrub browse species; **Grassland** = A station with less than 15 percent total canopy cover of shrub browse species; and **Riparian Forest** = A station located along a stream floodplain that is dominated by riparian/wetland tree species;

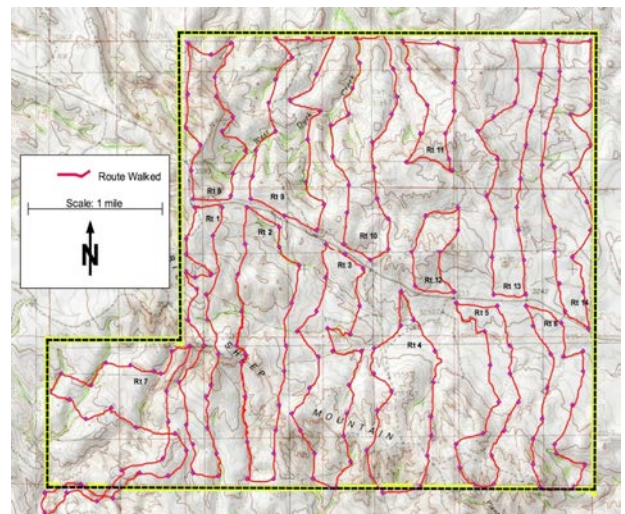
^eLandscape position is described in one of four broad categories that best characterizes the station. These broad groupings of landform type are reflected in vegetation potential and in animal habitat value. The terminology used in this study is defined as follows: **Slope** = A sloping or rolling terrain characterizes the overall length of the station, often with a complex combination of variable aspect and steepness. Slope is further described by categories of steepness and aspect; **Benchland** = Nearly level to gently sloping uplands, including low terraces that lack riparian vegetation, that are largely independent of aspect; **Riparian** = Low bottomland associated with a drainage-way that supports vegetation requiring a wetter moisture regime than is provided by surrounding uplands. The riparian landscape position is assumed to be independent of slope steepness and aspect; and **Broken topography** = Highly eroded and dissected landforms with large amounts of naturally occurring bare ground, such as is typically found in clay parent materials—also known as “badland topography.”

^fSlope steepness categories represent a simplified characterization of an often complex landform of a station, where: **Flat** = 0 to 5 percent slope; **Gentle** = 5 to 20 percent slope; **Moderate** = 20 to 50 percent slope; and **Steep** = Greater than 50 percent slope.

^gObserved fecal pellet frequency is recorded for each ungulate species by estimating the frequency of occurrence through each station, as: **High** = Pellet groups occurring along the station at intervals closer than 50 ft (15 m); **Moderate** = Pellet groups occurring along the station at intervals 50–100 ft (15–30 m); **Low** = Pellet groups occurring along the station at intervals greater than 100 ft (30 m); and **Limited** = No pellet groups observed.

In the study area, browse forage was not utilized evenly. Less palatable species were hardly browsed (for example western snowberry [*Symphoricarpos occidentalis*], creeping juniper [*Juniperus horizontalis*], and Plains silver sagebrush [*Artemisia cana* subsp. *cana*]), while almost every plant of the most preferred species was intensely browsed (for example chokecherry [*Prunus virginiana*], thorny buffaloberry [*Shepherdia argentea*], and western serviceberry [*Amelanchier alnifolia*]).

Figure 5 shows the occurrence distribution of thorny buffaloberry (*Shepherdia argentea*), a highly desired browse species. Thorny buffaloberry, even though armed with a thorny aspect, is intensely browsed throughout the study area. In all cases it was hedged down into the “arrested” growth form (Keigley and Frisina 1998), far shorter than the normal stature of the species. Seedlings or saplings of the species were not observed. These are indications that the population of this important species is in decline in the study area.

**Figure 3**—Map of the study area showing the layout of routes sampled.

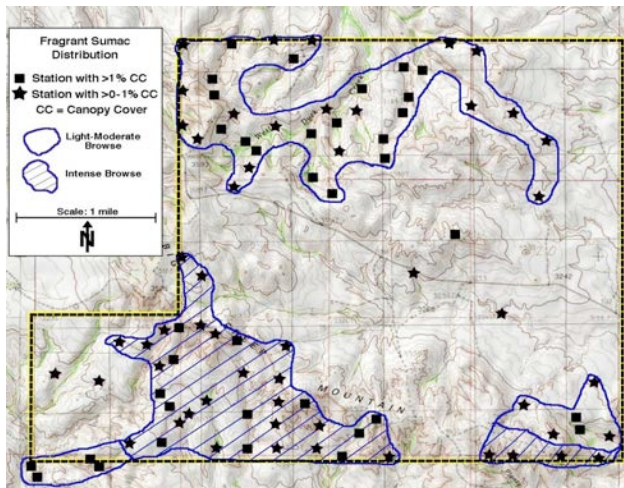


Figure 4—Approximate distribution of fragrant sumac (*Rhus aromatica*) occurrence on the study area, showing locations with different levels of browse intensity.

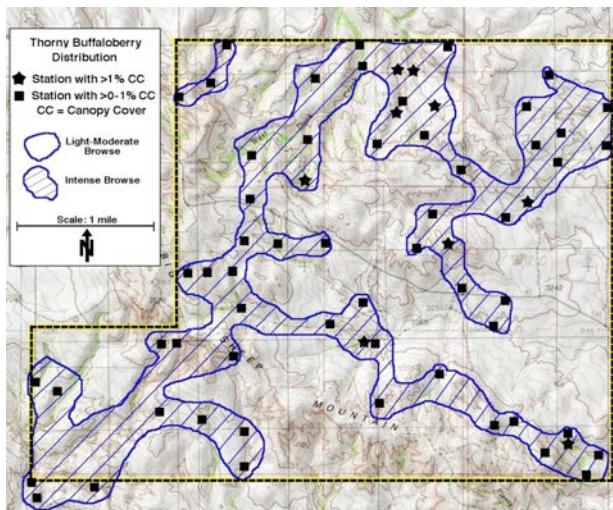


Figure 5—Approximate distribution of thorny buffaloberry (*Shepherdia argentea*) occurrence on the study area, showing locations with different levels of browse intensity.

Large Wild Ungulate Distribution (Pellet Group Frequency)

Estimation of the frequency of ungulate species fecal pellet groups was made by ocular observation for each station along the browse assessment routes walked, although white-tailed deer (*Odocoileus virginianus*) pellets were not distinguished from those of mule deer (*Odocoileus hemionus*). For wild ungulates known to use the site, pellet group frequency on each station polygon was categorized as:

- High—Pellet groups observed at a spacing closer than one per 50 ft (15 m) of route distance;
- Moderate—Pellet groups observed at a spacing of 50 to 100 ft (15 to 30 m) of route distance;
- Low—Pellet groups observed at spacing greater than 100 ft (30 m) of route distance; or
- Limited—No pellet groups observed on the station.

The distributions of pellet group frequencies recorded is tabulated in table 3. Although both mule deer (*Odocoileus hemionus*) and white-tailed deer (*Odocoileus virginianus*) are present, fecal pellets were not distinguished between the two deer species. Pronghorn antelope (*Antilocapra americana*) are present on the study area, but their fecal pellets were not observed along the routes sampled.

CONCLUSIONS

Who Did What?

This study was not designed to differentiate the impacts of the different ungulate species that utilize the range; nor was it designed to distinguish livestock impacts from those of wildlife. Questions about which animals are eating which plants may need to be answered before management decisions intended to bring habitat improvement can be made with much confidence. While both wildlife and livestock utilize all accessible portions of the range, very few deer pellets were found on some of the routes where severely browsed shrubs were observed. Relatively lush herbaceous growth in many parts of the study area may have obscured observation of fecal pellets.

Bang for the Buck

This effort was largely a test to find out how much area could be sampled by an observer in a given length of time, or for a certain dollar amount, and to what resolution the data could be interpreted. The study area encompassed 10 sections (6,400 ac [2,590 ha]) of fairly rugged terrain containing a diverse vegetation. Approximately 54 mi (86 km) of route were surveyed with 250 stations sampled, giving a linear resolution of approximately 0.18 mi (0.29 km) to the data.

Expenses to accomplish the field sampling and subsequent analysis and reporting were held within a \$10,000 budget for this study area. However, to cover the study area to the resolution specified, required approximately 50 percent more field time than was budgeted. Future applications of this methodology should figure the field time at 2.5 hours per mile of route. (This is to walk the routes and collect the data at a data intensity of five to six stations per mile.) Considering travel distance to remote sites, inclement weather delays, and the time to adequately make the observations, this initial work was budgeted approximately 25 percent too low.

Suggestions

Managers of wildlife populations and habitats may now evaluate this new tool for its potential to serve their needs and to integrate with their traditional methods. This method is intended for rapid, survey application, where spatially detailed information is needed across large areas. The test data collected is available for scrutiny, criticism, suggestion for improvement, and creative application to management needs. The mapped variable examples presented in this paper are from the full set included in the report to MT FWP (Thompson and Hansen 2007).

Table 2—Browse species recorded with canopy cover of each, the number of stations and percent of stations having the species recorded (constancy of occurrence) (total number of stations = 250).

Browse Species	Project wide avg. canopy cover ^a (%)	Range of canopy cover (%)	#Stations w/ species recorded	% Constancy of occurrence ^b
Western snowberry (<i>Symphoricarpos occidentalis</i>)	6.1	0-60	202	80.8
Green ash (<i>Fraxinus pennsylvanica</i>)	5.4	0-3	74	29.6
Common chokecherry (<i>Prunus virginiana</i>)	4.1	0-10	88	35.2
Creeping juniper (<i>Juniperus horizontalis</i>)	2.6	0-10	169	67.6
Silver sagebrush (<i>Artemisia cana</i>)	2.0	0-40	213	85.2
Fragrant sumac (<i>Rhus aromatica</i>)	0.8	0-10	90	36.0
Common juniper (<i>Juniperus communis</i>)	0.5	0-10	74	29.6
Woods rose (<i>Rosa woodsii</i>)	0.5	0-3	61	24.4
Arkansas rose (<i>Rosa arkansana</i>)	0.4	0-3	122	48.8
Thorny buffaloberry (<i>Shepherdia argentea</i>)	0.2	0-3	66	26.4
Western serviceberry (<i>Amelanchier alnifolia</i>)	0.2	0-3	43	17.2
Kinnikinnick (<i>Arctostaphylos uva-ursi</i>)	0.2	0-0.5	10	4.0
Common rabbitbrush (<i>Chrysothamnus nauseosus</i>)	0.1	0-10	35	14.0
Golden currant (<i>Ribes aureum</i>)	0.1	0-3	31	12.4
Missouri gooseberry (<i>Ribes setosum</i>)	0.1	0-0.5	26	10.4
Red-osier dogwood (<i>Cornus stolonifera</i>)	0.1	0-0.5	19	7.6
Succulent hawthorn (<i>Crataegus succulenta</i>)	0.1	0-3	17	6.8
Bebb willow (<i>Salix bebbiana</i>)	0.1	0-10	16	6.4
Great Plains cottonwood (<i>Populus deltoides</i>)	0.1	0-20	10	4.0
Water birch (<i>Betula occidentalis</i>)	0.1	0-10	6	2.4
Rocky Mountain juniper (<i>Juniperus scopulorum</i>)	<0.1	0-10	10	4.0
Yellow willow (<i>Salix lutea</i>)	<0.1	0-10	5	2.0
Silverberry (<i>Elaeagnus commutata</i>)	<0.1	0-3	3	1.2
Black currant (<i>Ribes americanum</i>)	<0.1	0-0.5	3	1.2
American plum (<i>Prunus americana</i>)	<0.1	0-3	2	0.8
Common hop (<i>Humulus lupulus</i>)	<0.1	0-3	2	0.8
Peachleaf willow (<i>Salix amygdaloides</i>)	<0.1	0-10	1	0.4
Black cottonwood (<i>Populus trichocarpa</i>)	<0.1	0-20	1	0.4
Wyoming big sagebrush (<i>Artemisia tridentata</i> subsp. <i>wyomingensis</i>)	<0.1	0-3	1	0.4

^aAverage canopy cover = sum of (station polygon canopy cover of species X station length) / sum of all stations lengths)

^bConstancy of occurrence = number of stations where a species was recorded / total number of stations on the project (250)

Table 3—Ungulate pellet group frequency recorded for the 250 stations in Study Area 2.

Pellet Group Frequency	Number of Stations by Ungulate Species	
	Deer	Pronghorn Antelope
High (<50 ft [15 m] apart)	1	0
Moderate (50–100 ft [15–30 m] apart)	10	0
Low (>100 ft [30 m] apart)	38	0
Limited (none observed)	201	250

REFERENCES

- Daubenmire, R.D. 1959. A canopy-coverage method of vegetation analysis. Northwest Science 33:43-66.
- Frisina, M., Knapp, S. 2006. Mule deer winter range landscape level habitat survey. In: Request for Proposal: Vegetation Evaluation in Moose and Mule Deer Habitat. Montana Department of Fish, Wildlife and Parks. Helena, USA. 5p.
- Hansen, P.L., Pfister, R.D., Boggs, K., Cook, B.J., Joy, J., Hinckley, D.K. 1995. Classification and management of Montana's riparian and wetland sites. Miscellaneous Publication No 54. Montana Forest and Conservation Experiment Station, School of Forestry, University of Montana, Missoula, Montana, USA. 646p.
- Keigley, R.B., Frisina, M.R. 1998. Browse evaluation by analysis of growth form. Volume 1 Methods for evaluating condition and trend. Montana Fish Wildlife and Parks. Helena, Montana, USA. 153p.
- Thompson, W.H., Hansen, P.L. 2007. A landscape level habitat survey of mule deer winter range on the Weaver Ranch (north-central Montana) and Big Sheep Mountain (eastern Montana). Prepared for Montana Department of Fish, Wildlife and Parks. Helena, Montana, USA. 114p.
- USDA Forest Service. 1993. Rangeland ecosystem analysis and management handbook. FSH 2209.21 U.S. Forest Service, Intermountain Region, Ogden, Utah, USA.

A Process to Monitor and Manage Ungulate Browsing Pressure

Richard B. Keigley¹ and Michael R. Frisina²

ABSTRACT

Plant communities are monitored to prevent ungulate browsing from having an adverse effect on habitat. The monitoring process described in this paper applies to plant species capable of growing through the browse zone, and uses two measurements (percent intensely browsed and LD Index) to determine if browsing will prevent the height growth of a young plant. To determine the potential effect of browsing on habitat structure, two variables are compared: 1) Existing browsing pressure is the percent of young plants that browsing is predicted to prevent from growing to full height. 2) Threshold browsing pressure is the percent of young plants that must grow to full height to attain a specified habitat structure. The process includes specifying target conditions, data collection and analysis, and refinement based on that analysis. The process requires a long-term commitment, but a minimal investment of time each year. By repeating the process over a period of years, the effects of browsing are documented and steps in the process are refined.

INTRODUCTION

It has long been recognized that ungulate browsing can have a major effect on communities of browse plants (Wright and Thompson 1935; Rasmussen 1941; Leopold 1943; Mussehl and Howell 1971; Hebblewhite and others 2005; Binkley and others 2006; Wagner 2006). To prevent ungulates from having an adverse effect on habitat, wild and domestic ungulate populations have been managed based on the monitoring of vegetation condition (Keigley and Frisina 1998; Mussehl and Howell 1971). This paper describes problems associated with a common method of monitoring (utilization rate), and describes a height-based approach that addresses those problems.

PROBLEMS WITH UTILIZATION-BASED MONITORING

Browse utilization by ungulates is measured in two ways: incidence of use (Cole 1959; Stickney 1966) and biomass consumed (Basile and Hutchings 1966; Schmutz 1983; Bilyeu and others 2007). Ungulate population objectives can be defined by specifying a maximum utilization rate such as 35 percent twigs browsed or biomass removed. Utilization values measured in the field are compared to target values to determine if the population objectives were attained.

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Richard B. Keigley, Northern Rocky Mountain Science Center, USGS, Bozeman, MT

²Michael R. Frisina, Montana Fish Wildlife and Parks, Butte, MT

The utilization-based approach has the following problems. First, the biologic effect of 35 percent twigs browsed may differ from the biologic effect of 35 percent biomass removed. The relationship between utilization and biologic effect must be determined separately for both measurement protocols. Second, utilization measurements may not accurately reflect the actual effect of browsing. Protracted heavy browsing can stress plants, causing them to produce very short lengths of current-year-growth. If ungulates avoid consuming those short lengths, the measured utilization rate could be quite low. The measured rate could indicate an appropriate level of use, when in fact the plant is stressed by browsing and may be dying. Third, the potential effect of excessive utilization is seldom specified. Those effects include premature death, decreased production, reduced recruitment, and decreased height growth. If the effect is unspecified, there is no way to determine if utilization at some level actually had an adverse effect. Fourth, there is no direct relationship between utilization rate and structural habitat characteristics. For example, there is no obvious link between browsing at a rate of 35 percent and the species composition, stem height, and stem density that might result in the future. Because there is no prior expectation as to what browsing at 35 percent utilization will produce, a manager will likely accept whatever condition results. However, there is no guarantee that those conditions will be suitable for meeting land management objectives.

The height-based approach described below addresses the problems described above. However, the approach can only be applied to tall-growing trees and shrubs. In the case of short-growing species such as sagebrush, saltbush, and winterfat, monitoring by utilization may remain the best alternative.

A HEIGHT-BASED APPROACH

The height-based approach is a refinement of methods described in Keigley (1997), Keigley and Frisina (1998), Keigley and others (2002ab), Keigley and Fager (2006). The approach focuses on the effect that ungulates can have on plant height growth. It is best applied to species that, under light browsing, are capable of growing through the browse zone. Heavy browsing can prevent young plants of such species from growing taller than 30 to 50 cm (Keigley and Frisina 2008). If heavy browsing continues over a prolonged period of time, a meadow-like condition can be produced as the older, taller shrubs die, while browsing prevents the height growth of young plants.

The approach is based on two premises. The first premise: The structural characteristics of the plant community are of primary importance. Those structural characteristics include species composition, stem density, and diversity of stem heights. The attainment of appropriate habitat structure is critical to attaining land management objectives. The second premise: Browsing should not be the dominant factor that influences the structure of the plant community.

In this paper we define our use of eight terms that are part of a systematic process. The aim of the process is to determine if browsing will affect the future structure of the plant community by preventing young plants from growing to full stature. The terms are described below. An example of the process is described in the final section.

DEFINITION OF TERMS

Indicator Species

A given plant community will likely contain several browse species, and likely it would not be feasible to measure the effect of browsing on each. Fortunately a single, or relatively few, species can be used to describe the approximate effect of browsing on all other browse species. Three criteria guide the selection of indicator species: height, presence of bud scars, and preference.

Because we focus on the effect of browsing as the plant grows within the browse zone, we restrict our attention to species in the community that are capable of growing to 2.5 m tall, or at least significantly into the browse zone. In addition, it must be possible to determine annual growth increments using bud scars; species with naked buds cannot be used.

Ungulates prefer some browse species more than others. For example, Bebb willow, Geyer willow, alder, and spruce form a preference gradient that runs from highly-preferred to least-preferred. When browsing increases, the more-highly-preferred species are heavily consumed before less-preferred species. If the increase occurred in a community that contained the above species, Bebb willow would be first to be heavily used. At this point, Geyer willow might be lightly used; alder and spruce might not be used at all. A continued increase in browsing would cause heavy use of Geyer willow, with alder and spruce lightly used. Alder would be next, and finally spruce. We infer the existence of this preference gradient because we can find examples of the combinations described above.

By selecting an appropriate indicator species, we can accomplish the following. First, if we select a highly preferred species such as Bebb willow, and find its condition to be acceptable with respect to browsing, we can assume that the condition of all other species is acceptable. We will have documented that current browsing has no effect on habitat structure. Second, we could use Bebb

willow to document the earliest stage of habitat decline. Third, if habitat quality is in further decline, we can measure the magnitude of that decline by determining which species are heavily used and which are not. Are we at the point where Geyer willow, but not alder is heavily browsed? Finally, we can document changes in habitat condition that occur due to management action. Species at the beginning stages of heavy use will also be most sensitive to changes in browsing pressure. For example, if alder is heavily browsed (but not spruce), we can expect a reduction in browsing pressure to be reflected by the growth of alder before we see a response in Geyer willow.

Potential Stature

Potential stature is a concept used to distinguish the effect of browsing from other factors that may influence height growth. The concept allows us to deal with situations where, for reasons unrelated to browsing, plants do not grow to their expected height. In the absence of browsing, the mature individuals of a plant community grow to a stature that is determined by local conditions including climate, soil, slope, aspect, disease, and insect herbivory. We include in this list everything but browsing. We could refer to this height as the *potential stature in the absence of browsing at this local site*, or more simply as “potential stature.”

Below we describe a criterion that marks the prevention of height growth by browsing. If that criterion is met, the plant is deemed unlikely to grow to potential stature. Otherwise, the plant is deemed likely to grow to potential stature. We do not need to know the actual potential height in m.

The concept of potential stature allows us to deal with situations where unfavorable environmental or physiologic conditions limit height growth. As long as the abovementioned criterion is not met, the plant is assumed to be growing to its potential stature even though unfavorable conditions prevent it from growing out of the browse zone. For example, an unbrowsed aspen sucker that dies from shading is presumed to have grown to its potential stature.

Browsing Level

Browsing level is the first of two attributes used to determine if browsing will prevent a given plant from growing to its potential stature; LD Index—described below—is the second. Browsing level is based on morphologic evidence, and provides a means of assigning a plant into one of two categories: 1) Light-to-moderate (browsing will probably not prevent growth to potential stature), and 2) Intense (browsing may prevent growth to potential stature). If the morphologic evidence suggests that browsing may prevent growth to potential stature, LD Index is used to determine if that is likely the case; this is the criterion mentioned in the preceding section.

The classification of browsing level is based on whether or not browsing caused the death of a complete annual increment of any vertically oriented stem. If a relatively small length of stem is removed by browsing, current-year-growth the following year will likely develop from a lateral bud on the browsed segment that was current-year-growth the previous year. The development of current-year-growth on the previous-year-segment allows part of that segment to remain alive. If this condition occurred throughout the life of the stem, it is classified as light-to-moderately browsed (figure 1).

If a substantial length of a current-year-growth segment is consumed, the entire segment may die (figure 2). Death occurs because no current-year-growth develops from that segment the following year. To be classified as intensely browsed, two criteria must be met: 1) a complete vertically-oriented annual increment must be dead, and 2) the increment must have been browsed. The death of a complete vertically-oriented annual increment is an interruption in height growth. LD Index is used to determine if browsing will likely prevent further height growth.

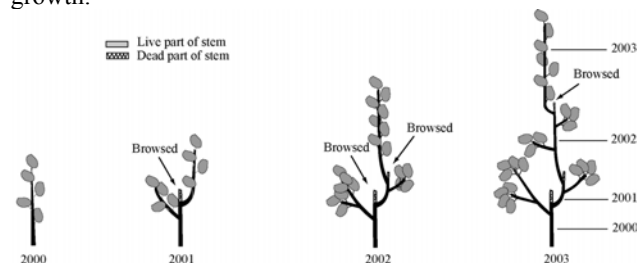


Figure 1—Light-to-moderately browsed stems. Browsing is classified by examining vertically oriented stems. These drawings show how the plant might appear from 2000 to 2003. Although the plant was browsed each year, part of each year's annual segment remained alive.

Once a stem is classified as intensely browsed, it retains that classification for the rest of its life. The browsing level of a plant is determined by examining its stems. Trees typically have a single stem. However browsing, as well as other influences, may cause the vertical growth of multiple stems. If any vertically oriented stem is intensely browsed, the tree is classified as intensely browsed. By definition, a shrub consists of multiple vertically oriented stems. If any of the shrub's vertically-oriented stems were intensely browsed (including stems that are completely dead), the shrub is classified as having been intensely browsed. Once a plant is classified as intensely browsed, it retains that classification for life.

LD Index

LD Index (Live-Dead Index) is a measure of height growth that occurred after a plant was intensely browsed (figure 3). It is based on two measurements, the first being the height to the base of the tallest current-year-growth (H_{BCYG}). (All measurements are made to the base of current-year-growth

to minimize complications that arise if a stem is browsed after measurement.) The second measurement is the height to the tip of the tallest stem having a dead, vertically oriented, complete annual increment that was browsed (H_D ; see figure 2).

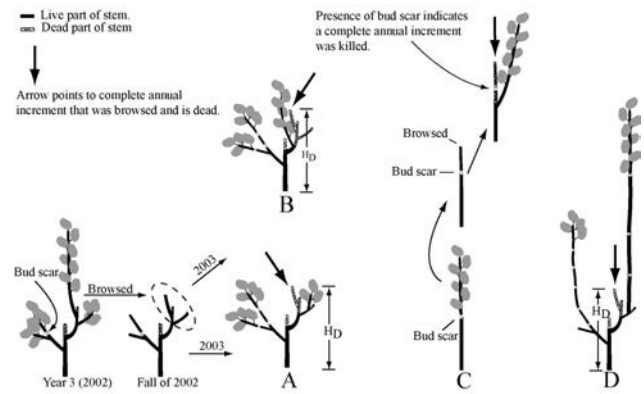


Figure 2—Intensely browsed stems. The left side of the figure depicts an aspen as it might appear during the 2002 growing season, and after browsing in the fall; we follow the fate of the browsed leaders (encircled by the ellipse) that developed from lateral buds. In Figure 2A, the upper leader died and the lower leader survived. In Figure 2B both the upper and lower leaders died. In Figure 2C the leader developed from a terminal bud; to identify complete annual increments we must inspect stems for the presence of terminal bud scars. In Figure 2D extensive growth occurred after the stem was intensely browsed. We continue to classify the stem as intensely browsed; LD Index is used to determine if browsing is likely to prevent height growth. The height to the tip of the tallest dead stem increment (H_D) is a measurement used to calculate the stem's LD Index.

An LD Index of about zero (figure 3a) indicates that current-year-growth is being browsed to the level of mechanical protection provided by stems and twigs that were killed by browsing. An LD Index of about zero indicates that browsing is preventing height growth.

If the LD Index is substantially less than zero (figure 3b), the plant is dying back to ground level. In the case of trees, the death of the stem at ground level means the death of the plant. Because shrubs can produce additional stems from belowground, the plant will persist as long as additional stems can be produced. The plant diagramed in figure 3b illustrates a succession of such stems. An LD Index of substantially greater than zero (figure 3c) indicates the plant is growing taller after having been intensely browsed.

How Browsing Level and LD Index are Used

We make the assumption that browsing will not prevent the height growth of a light-to-moderately browsed plant. This assumption can be tested as part of the systematic process described below. To determine if browsing will prevent the height growth of an intensely browsed plant, we compare its measured LD Index value to a threshold LD Index that,

initially, is estimated. If the plant's measured LD Index value is greater than or equal to the threshold value, we assume that browsing will not prevent height growth. If the LD Index is less than the threshold value, we assume that browsing will likely prevent height growth. This is the criterion that marks a failure to grow to potential stature due to browsing. An initial threshold LD Index value of 50 cm is a reasonable starting point. As monitoring at a site continues over a period of years, data are collected to adjust the threshold LD Index upward or downward. The process for evaluating threshold LD Index is described in Step 5 of the last section of this paper.

Browsing-related Architectures

The degree to which a plant is browsed affects its shape. We refer to those shapes as browsing-related architectures. Four browsing-related plant architectures document the effect of browsing as plants grow through the browse zone (Keigley 1997; Keigley and Frisina 1998; Keigley and others 2002a). The uninterrupted-growth-type architecture is produced by light-to-moderate browsing; the retrogressed-type architecture is produced by an increase from light-to-moderate browsing to intense browsing; the arrested-type architecture is produced by intense browsing; the released-type is produced by a change from intense browsing to light-to-moderate browsing. The architectures apply to both trees and shrubs. Figure 4 illustrates the architectures as expressed by aspen. The browsing related architectures are used to conduct a rapid survey of large areas to select indicator species and identify sites where monitoring should be a priority.

Replacement Percentage

Replacement percentage is the first of three concepts used to relate ungulate browsing to its effect on plant community structure—the other two are existing browsing pressure and

threshold browsing pressure. For a stand of plants to be sustained, young plants must grow through the browse zone to replace those that die. Replacement percentage is the percent of young plants that must grow through the browse zone to produce the stem density of the parent stand.

Replacement percentage varies by site and is influenced by the lifespan of the species and the age structure of the stand. Where young plants replace plants that die of old age, replacement can be largely incremental. On an annual basis, the incremental replacement rate could be very low. The highest replacement rate occurs when all stems must be simultaneously replaced due to an event such as fire.

For two reasons, we use simultaneous replacement as the basis for assessing the potential effect of browsing on habitat structure. First, the determination of simultaneous replacement percentage is much less complicated than the determination of incremental replacement percentage, and for that reason more appropriate for general monitoring. Second, all sites are potentially exposed to wildfire or prescribed fire. We base the determination of replacement percentage on the hypothetical occurrence of fire.

Replacement percentage is calculated using two estimates of stem density. The first is the stem density that we desire to grow through the browse zone after a hypothetical fire. This is the target stem density. The second is the density of stems that will be available after that hypothetical fire to grow through the browse zone and establish the target stem density. This is the propagation stem density. Depending on the species, post-fire propagation could be from seed, root suckers, rhizomes, or root crown. Each of these sources is a propagule. The propagation density is determined by estimating the density of propagules that would be present after a hypothetical fire.

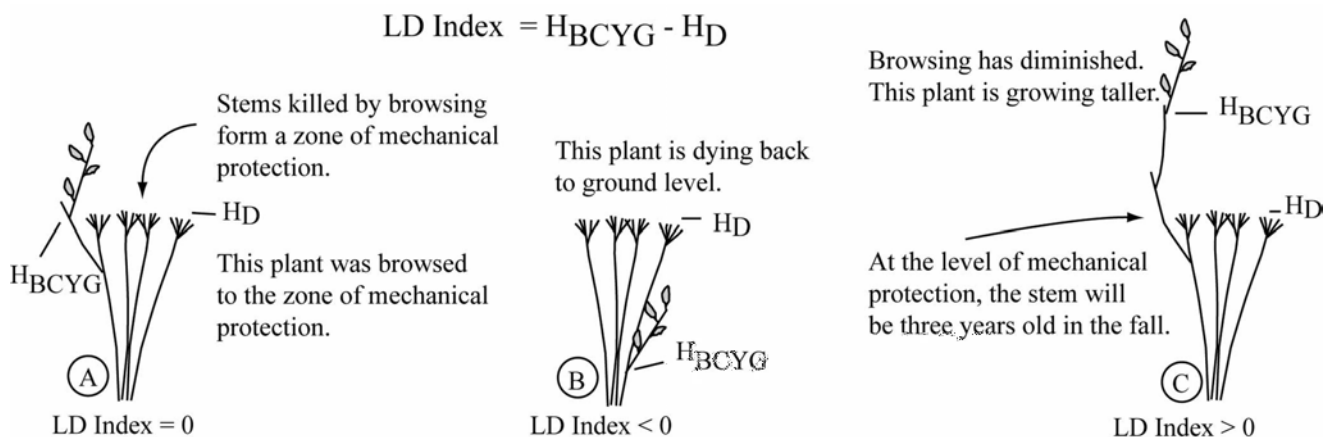


Figure 3—Measurement of LD Index. LD Index is calculated by subtracting the height of the tallest stem having a complete dead annual increment (H_D) from the height of the tallest live leader as measured to the base of current-year-growth (H_{BCYG}).

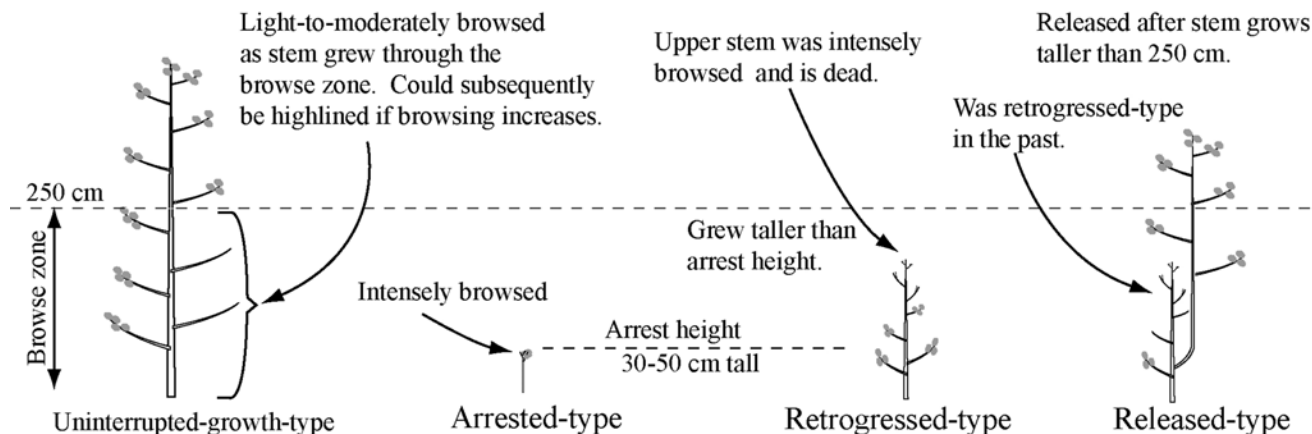


Figure 4—Browsing-related architectures as expressed by trees. The uninterrupted-growth-type architecture is produced by light-to-moderate browsing as the trunk grows through the browse zone. Arrested-type architecture is produced by the chronic intense browsing of young plants. Retrogressed-type architecture is produced by a change from light-to-moderate browsing (that allowed the plant to grow taller than arrest height) to intense browsing (that prevented further height growth). Released-type architecture is produced by a change from intense browsing to light-to-moderate browsing.

Replacement percentage is calculated by dividing the target stem density by the propagation stem density. In the case of aspen, the number of suckers present after a fire may be substantially greater than the number required to produce the desired future stem density; the replacement percentage could be much less than 100 percent. To re-establish a mature stand of willow after fire, it may be necessary for stems from each root crown to grow through the browse zone; the replacement percentage could be 100 percent. In the following section, replacement percentage is used to determine the point at which ungulate browsing will be a dominant factor in shaping the structure of the plant community.

Browsing Pressure: Existing and Threshold

We define browsing pressure to be the percent of plants that are prevented by browsing from attaining their potential stature. A browsing pressure of 100 percent means that browsing prevents all plants from attaining potential stature. There are two types of browsing pressure: the existing browsing pressure (which is measured), and the threshold browsing pressure (which is calculated).

In a previous section we described how browsing level and LD Index could be used to determine if browsing will prevent an individual plant from growing to potential stature. To determine the effect of browsing on the future structure of the plant community, we examine a sample of 20 plants that are between 50 and 150 cm tall. Plants in this size range are attempting to grow through the browse zone to replace plants that die of old age or for other reasons. Such plants are tall enough to have been exposed to browsing, and are short enough to be available to all ungulate species.

Based on the sample of 20 plants, we estimate the percent that browsing will prevent from growing through the browse zone. This is the percentage of plants having a measured LD Index that is less than the threshold value. We refer to that percentage as the existing browsing pressure.

Threshold browsing pressure describes the point at which browsing will affect stand structure. It is calculated by subtracting the replacement percentage from 100 percent. For example, if the estimated replacement percentage of a stand of willow was 100 percent, the threshold browsing pressure would be 0 percent.

To determine the effect of browsing, we compare the existing browsing pressure to the threshold browsing pressure. If the existing browsing pressure is less than (or equal to) the threshold value, then browsing will not have a dominant effect on the structure of the plant community. If the existing browsing pressure is greater than the threshold value, then browsing is predicted to have a dominant effect on stand structure.

MONITORING PROGRAM: AN EXAMPLE

Using the browsing-related architectures an indicator species is selected, and one or more sites are identified as permanent monitoring sites. At each monitoring site, the systematic process described in figure 5 is conducted. To be successful, a commitment should be made to monitor each site for a period of decades. This will allow a succession of managers to compare predicted results with actual results, and make corrections as appropriate. Establishing a site and collecting the first year's data should require about four hours. Once a site is established, the time required for monitoring should require one to two hours; annual monitoring may not be required. Data analysis can be automated in a spreadsheet.

Step 1: Describe the desired target density of stems that are taller than 2.5 m.

The target density is the density of stems we would like to grow through the browse zone after a hypothetical fire. It is a rough approximation that could be based on measurements taken at an existing stand, or could be based on past experience. In this example we used aspen data collected at the Mt. Fleecer Wildlife Management Area in southwest Montana. Using a belt transect we determined that the density of stems taller than 2.5 m was 0.2 stems / m². A desired density could also have been estimated based on an approximate spacing of stems. Stems spaced 2.2 m apart result in a density of 0.2 stems / m².

Step 2: Measure the existing density of stems that are greater than 2.5 m tall and the density of stems that are less than 2.5 m tall.

As described in Step 1, the density of stems taller than 2.5 m was 0.2 stems / m². The density of stems shorter than 2.5 m was 1.0 stems / m². We use this density as an estimate of the propagation density (suckers that would be present after fire) recognizing that an actual fire could affect the production of suckers. When the process is repeated in subsequent years, the measurements taken in Step 2 provide the basis for comparing the predicted result to the actual result, and for making adjustments accordingly.

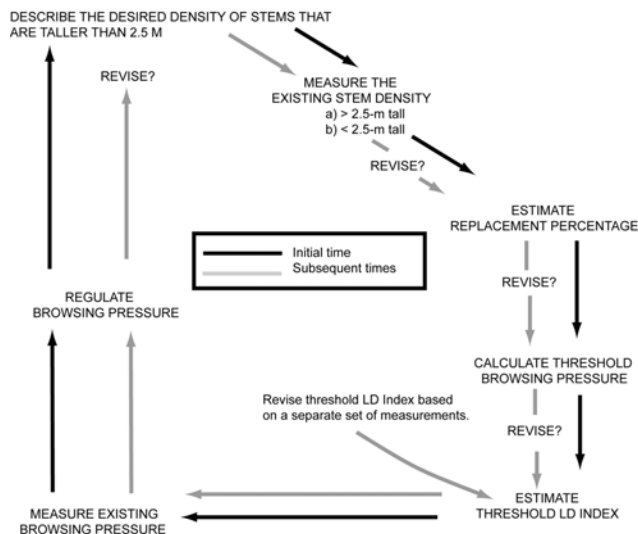


Figure 5—Flowchart of the process. The process consists of a series of steps aimed at regulating browsing to attain specific habitat characteristics.

Step 3: Estimate the replacement percentage.

To calculate replacement percentage we divide the target density by the propagation density: 0.2 stems per m² / 1.0 stems per m² = 20 percent. Based on our current information, if all of the tall stems were to simultaneously die, the existing stand density would be attained if 20 percent of the short stems were to grow through the browse zone.

When calculating replacement percentage, the propagation density is based on how the species propagates after fire. In the case of aspen, stems are classified into the two categories based on their actual measurements. If a tall aspen is burned, that stem dies; recruitment only occurs due to the growth of suckers. Above, the density of those short stems was used as the propagation density. In the case of sprouting species, new stems could originate from plants of any height. In those cases, the propagation density would be based on the presence of plants of any height.

Step 4: Calculate the threshold browsing pressure.

We calculate the threshold browsing pressure by subtracting the replacement percentage from 100 percent: 100 percent - 20 percent = 80 percent. If browsing prevented 80 percent of the short stems from growing through the browse zone (which are present at 1 stem per m²), 20 percent would do so, theoretically resulting in a stand density of 0.2 stems per m². Other factors such as disease might influence the establishment of the stand, but as long as the existing browsing pressure is less than 80 percent, browsing will not be the dominant influence.

Step 5: Estimate a threshold LD Index.

In this example the initial estimated threshold LD Index was set at 50 cm, meaning that plants having an LD Index greater than (or equal to) 50 cm are predicted to grow through the browse zone. Data are collected to evaluate this estimate. A transect is established in which LD Index of 20 plants is periodically re-measured. (This is one of the two kinds of transects that would be established at each site.) It will likely be found that plants having small LD Indexes fail to grow taller over the measurement period, while plants initially having larger LD Indexes did grow taller. The smallest LD Index in the latter category would suggest a threshold LD Index.

Step 6: Measure the existing browsing pressure.

Existing browsing pressure is measured by sampling 20 individuals at equal intervals along a paced transect (this is the second type of transect established at each site). At each point the nearest plant 50 to 150 cm tall is measured. If plants in this range are absent (or some distance away), intensely browsed plants that are less than 50 cm tall can be included. Short light-to-moderately browsed plants are avoided because the absence of browsing may be due to their short stature. Plants are not necessarily re-sampled from one year to the next. The existing browsing pressure at a site is equal to the percent of plants that have an LD Index that is less than the threshold LD Index value.

In 2003 all aspen at the Fleecer site were intensely browsed; the maximum measured LD Index was 17 cm. Because all 20 plants had an LD Index that was less than the threshold LD Index of 50, the existing browsing pressure at this site in 2003 was 100 percent.

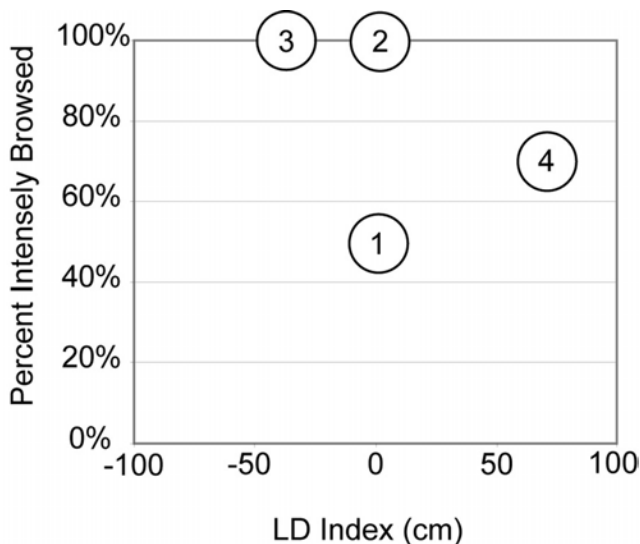


Figure 6—Graph of percent intensely browsed and LD Index. This graph provides a means of describing changes in a stand that may occur over time, or to describe differences between separate stands. Two circumstances could cause a stand to have the characteristics shown at point 1. First, an increase in browsing has prevented the height growth of some plants while others remain light-to-moderately browsed. Second, a decrease in browsing has allowed some young plants to grow into the browse zone; plants that were previously intensely browsed remain alive but fail to grow. The characteristics at Point 2 depict a heavily browsed community that is on the verge of entering decline. Point 3 depicts the characteristics of a community that is in decline. The characteristics depicted at Point 4 could be produced by a decrease in browsing that permitted plants that were previously intensely browsed to grow taller. A community of light-to-moderately browsed plants would not be plotted on this graph.

A graph of mean percent intensely browsed and mean LD Index can be used to describe changes over time or differences between sites (figure 6). The data collected in 2003 would be plotted slightly to the left of point 2. As described above, aspen was heavily browsed in 2003. We collected data at this site again in 2008. Those data would be plotted somewhat below point 1. A graph such as that shown in figure 6 could provide a focus for describing the potential reasons why the existing browsing pressure of aspen at this site was reduced from 100 percent in 2003 to 35 percent in 2008. Such a graph could also be used to describe how browsing might vary spatially.

Step 7: Regulate browsing pressure.

To attain or maintain the desired habitat, browsing pressure must be regulated so that existing browsing pressure is less than the threshold browsing pressure. A detailed discussion of the regulation of browsing pressure is beyond the scope of this paper. In part, ungulate population size is a factor. Climate can indirectly cause decreases or increases in browsing pressure by respectively making available more

or less of the browse resource. Declines in the abundance or vigor of a preferred browse species could lead to an apparent increase in browsing pressure on that species.

Repeat the process in subsequent years.

Using a set of approximations, the height based approach directly relates browsing to community structure. The data collected in the initial year provide a starting point. There will likely be a discrepancy between the anticipated result and actual result. This discrepancy can be reconciled by periodic adjustments based on periodic measurements. For example, in subsequent years the initial estimates of densities should be examined; it should be possible to improve the accuracy of threshold LD Index as a predictor of height growth. The long-term application of this systematic process should make it possible to determine if the relationship between ungulates and habitat is consistent with meeting land management objectives.

REFERENCES

- Basile, J.V., Hutchings, S.S. 1966. Twig diameter-length-weight relations of bitterbrush. *Journal of Range Management* 19:34-38.
- Bilyeu, D.M., Cooper, D.J., Hobbs, N.T. 2007. Assessing impacts of large herbivores on shrubs: Tests of scaling factors for utilization fates from shoot-level measurements. *Journal of Applied Ecology* 44:168-175.
- Binkley, D., Moore, M.M., Romme, W.H., Brown, P.M. 2006. Was Aldo Leopold right about the Kaibab deer herd? *Ecosystems* 9:227-241.
- Cole, G.F. 1959. Key browse survey methods. *Association of State Game and Fish Commissioners* 39:181-186.
- Dasmann, W.P. 1948. A critical review of range survey methods and their application to deer range management. *California Fish and Game* 34: 180-207.
- Hebblewhite, M., White, C.G., Nietvelt, C.G., McKenzie, J.A., Hurd, T.E., Fryxell, J.M., Bayley, S.E., Paquet, P.C. 2006. Human activity mediates a trophic cascade caused by wolves. *Ecology* 86:2135-2144.
- Kay, C.E. 1990. Yellowstone's northern elk herd: A critical evaluation of the "Natural Regulation" paradigm. PhD Dissertation. Utah State University, Logan, Utah.
- Keigley, R.B. 1997. A growth form method for describing browse condition. *Rangelands* 19:26-29.
- Keigley, R.B., Fager, C. 2006. Habitat-based adaptive management at Mount Haggin Wildlife Management Area. *Alces* 42:49-54.
- Keigley, R.B., Frisina, M.R. 1998. Browse evaluation by analysis of growth form. Volume 1: Methods for evaluating condition and trend. Montana Fish, Wildlife and Parks, Helena, MT.
- Keigley, R.B., Frisina, M.R. 2008. Aspen height, stem-girth and survivorship in an area of high ungulate use. *Northwest Science* 82: 199-210.
- Keigley, R.B., Frisina, M.R., Fager, C. 2002a. Assessing browse trend at the landscape level / Part 1: Preliminary steps and field survey. *Rangelands* 24:28-33.
- Keigley, R.B., Frisina, M.R., Fager, C. 2002b. Assessing browse trend at the landscape level / Part 2: Monitoring. *Rangelands* 34-38.
- Leopold A. 1943. Deer irruptions. *Wisc Conserv Bull* 8:3-11. Mussehl, T.W., Howell, F.W. 1971. Game management in Montana. Montana Fish and Game Department, Helena MT.
- Rasmussen, D.I. 1941. Biotic communities of the Kaibab Plateau, Arizona. *Ecological Monographs* 11:230-275.
- Schmutz, E.M. 1983. Browse class method of estimating shrub utilization. *Journal of Range Management* 36:632-637.
- Stickney, P.F. 1966. Browse utilization based on percentage of twig numbers browsed. *The Journal of Wildlife Management* 30:204-206.
- Wright, G.M., Thompson, B.H., 1935. Wildlife management in the national parks; Fauna Series No. 2, National Park Service. Government Printing Office, Washington, D. C., viii, 142 pp.

Getting The Biggest Bang For Your Buck: Wildlife Monitoring On Shrublands Of The Nevada Test Site

Derek B. Hall¹ and Paul D. Greger¹

ABSTRACT

The Nevada Test Site (NTS) covers 3,561 km² and extends over portions of both the Mojave and Great Basin Deserts. The resulting diverse and complex flora and fauna exhibit elements of both deserts. There are 20 vegetation associations, composed primarily of shrubs, nested within 10 vegetation alliances. Of the more than 1,200 invertebrate and 339 vertebrate species found in these shrubland habitats, 267 are considered sensitive or protected/regulated by federal or state laws. Wildlife and wildlife habitat monitoring ensures NTS activities comply with all federal and state laws enacted for the protection of these valuable biological resources and provides ecological information that can be used to predict and evaluate the potential impacts of proposed projects and current activities on these resources. This paper describes the monitoring approach used at this large site. Monitoring strategies include conducting preactivity surveys, proactively monitoring sensitive species, monitoring long-term population trends, and collaborating with other agencies and biologists. Ways to make monitoring more efficient and examples of successful monitoring and collaborations are discussed.

INTRODUCTION

The Nevada Test Site (NTS) is situated over portions of both the Mojave and Great Basin Deserts. Elements of both deserts are found in a diverse and complex flora and fauna. A transitional area (transition region) between the two deserts is also present. There are 20 vegetation associations nested within 10 vegetation alliances composed primarily of shrubs. Over 750 vascular plant species have been identified (Wills and Ostler 2001) of which 44 are sensitive and protected/regulated (National Security Technologies, LLC [NSTec 2008]) based on Nevada Natural Heritage Program's (NNHP) Animal and Plant At-risk Tracking List. More than 1,200 invertebrate and 339 vertebrate species are found in these shrubland habitats including one tortoise, 16 lizard, 17 snake, 241 bird, 44 terrestrial mammal, and 15 bat species (Wills and Ostler 2001). Of these, 267 are considered sensitive or protected/regulated (NSTec 2008) by federal or state laws including the desert tortoise (*Gopherus agassizii*) which is designated as threatened under the Endangered Species Act, 236 bird species protected under the Migratory Bird Treaty Act, and numerous other species protected by Nevada state law (for

example, game and furbearer species, rare or sensitive species) or other federal laws (for example, wild horses and burros).

Wildlife and habitat monitoring is required to (1) protect wildlife and their habitat, (2) ensure NTS activities comply with all federal and state laws, (3) keep NTS open for business, and (4) be environmentally responsible. This paper describes the monitoring approach used at this large site, discusses ways to make monitoring more efficient, and highlights some examples of successful monitoring and collaborations.

STUDY AREA

The NTS is administered by the U.S. Department of Energy, National Nuclear Security Administration Nevada Site Office (NNSA/NSO). The NTS is located in south-central Nevada, approximately 105 km northwest of Las Vegas, and encompasses approximately 3,561 km² (figure 1). Despite drastic changes to localized areas of the NTS due to nuclear testing activities for more than 40 years, biological resources over much of the NTS remain relatively pristine and undisturbed. NNSA/NSO estimates that only 7 percent of the site has been disturbed (U.S. Department of Energy, Nevada Operations [DOE/NV] 1996).

The southern two-thirds of the NTS is dominated by three large valleys or basins: Yucca, Frenchman, and Jackass Flats. Mountain ridges and hills rise above sloping alluvial fans to enclose these basins. Pahute and Rainier Mesas and Timber and Shoshone Mountains occupy the northern, northwestern, and west central sections of the NTS. Elevation on the NTS ranges from less than 1,000 m above sea level in Frenchman and Jackass Flats to about 2,340 m on Rainier Mesa.

The NTS has a climate characteristic of high deserts with little precipitation, hot summers, mild winters, and large diurnal temperature ranges. Monthly average temperatures in the NTS area range from 7°C in January to 32°C in July (DOE/NV 1996). The average annual precipitation on the NTS ranges from 15 cm at the lower elevations to 23 cm at the higher elevations (DOE/NV 1996). Most of the precipitation is received between December and March in the form of rain or snow with lesser amounts of rain usually received during July and August.

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Derek B. Hall and Paul D. Greger, National Security Technologies, LLC, Las Vegas, NV.

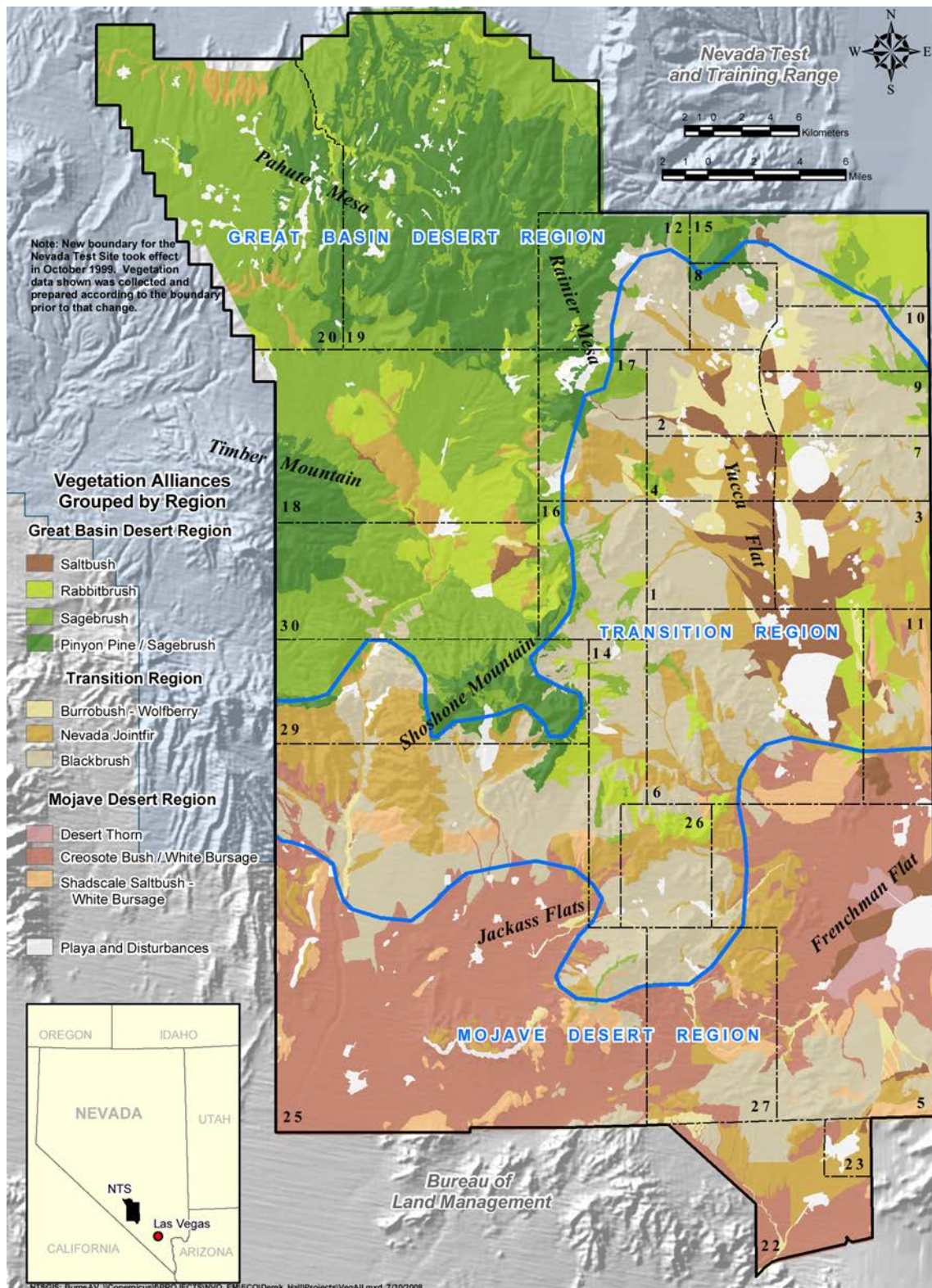


Figure 1—Location, major topographic features, and vegetation alliances and regions of the Nevada Test Site.

The NTS is located in an area of southern Nevada that lies between the Great Basin Desert and the Mojave Desert as defined by Jaeger (1957). Within the site boundaries are found both of these desert types. Transitional areas between the two deserts are also present having been created by gradients in precipitation, elevation, temperature, and soils. Unique combinations of physical site conditions have resulted in several different vegetation communities (Ostler and others 2000). Based on these vegetation communities, three distinct vegetation regions occur on the NTS: the Great Basin Desert, Mojave Desert, and Transition regions (figure 1). The Great Basin Desert region is a cold desert with dominant plant species consisting of sagebrush species (*Artemisia* spp.), singleleaf pinyon (*Pinus monophylla*), and Utah juniper (*Juniperus osteosperma*) and includes 10 associations nested within four alliances (41 percent of NTS). The Mojave Desert region is a hot desert with dominant plant species being creosote bush (*Larrea tridentata*) and white bursage (*Ambrosia dumosa*) with three associations nested within three alliances (southern 22 percent of NTS). The Transition region is transitional between the Great Basin and Mojave Desert regions with dominant plant species consisting of blackbrush (*Coleogyne ramosissima*), Nevada jointfir (*Ephedra nevadensis*), and burrobrush (*Hymenoclea salsola*) and includes seven associations nested within three alliances (37 percent of NTS) (Ostler and others 2000).

MONITORING APPROACH

The Ecological Monitoring and Compliance Program (EMAC) is funded by the U.S. Department of Energy, National Nuclear Security Administration, Nevada Site Operations Office, and is designed to ensure compliance with applicable laws and regulations, delineate and define NTS ecosystems, and provide ecological information that can be used to predict and evaluate the potential impacts of proposed projects and current activities on those ecosystems. Wildlife and habitat monitoring is a large part of EMAC. Monitoring serves several purposes including (1) protecting important resources (for example, desert tortoise clearance and preactivity surveys), (2) identifying human health and safety risks (for example, West Nile Virus, mountain lion or puma [*Puma concolor*] attacks), (3) defining species distributions (for example, fairy shrimp, skinks [*Eumeces* spp.]) and sampling in previously non-sampled areas), (4) identifying important habitat features (for example, bat roost sites and raptor nest sites), (5) assessing long-term population trends (for example, mule deer [*Odocoileus hemionus*]), and (6) detailing natural history and ecology information (for example, western burrowing owls [*Athene cunicularia hypugaea*]). Habitat monitoring is also conducted to document long-term vegetation changes and evaluate revegetation success in desert tortoise habitat, after large wildland fires, on revegetated cleanup sites, and on vegetated waste cover caps. Restoring wildlife habitat is a major goal of revegetation efforts.

While all species are inherently valuable to the ecosystems in which they are found, certain species have been afforded legal protection or are more vulnerable to becoming threatened or endangered than other species due to their rarity, economic value, or threats to their existence. These species make up the NTS Sensitive and Protected/Regulated Species List and have monitoring priority based on "Guiding Principles and Prioritization Criteria for Ecological Monitoring at the Nevada Test Site" and the NTS Resource Management Plan (DOE/NV 1998). Other species are also monitored to fill in data gaps or when deemed necessary. A sensitive species is defined as a species that is on the NNHP Animal and Plant At-risk Tracking List or designated as having a High or Moderate ranking in the Western Bat Species: Regional Priority Matrix (Western Bat Working Group 1998). A sensitive species is considered sensitive because it is rare or has some level of imperilment. A Protected/Regulated species is a species that is protected and/or regulated by federal or state law for various reasons (for example; rarity, economic value, cultural icon). For example, all but five bird species that occur on the NTS are protected under the Migratory Bird Treaty Act but relatively few of them are rare or in danger of becoming threatened or endangered. Some species are considered both sensitive and protected/regulated (for example, spotted bat [*Euderma maculatum*], desert tortoise, LeConte's thrasher [*Toxostoma lecontei*]).

The Sensitive and Protected/Regulated Species List is reviewed annually and updated as necessary. Several sources are checked to determine if changes to the list are warranted including the NNHP Animal and Plant At-risk Tracking List, U.S. Fish and Wildlife Service (FWS) threatened and endangered species website, and Nevada Administrative Code 503. Additionally, NTS biologists are involved in working groups and professional societies (for example, Nevada Native Plant Society, Nevada Bat Working Group and Western Bat Working Group) that largely determine the status of certain plant and animal species in Nevada.

The main strategies used to monitor wildlife and their habitat on the NTS include (1) conducting preactivity surveys and desert tortoise clearance surveys, (2) conducting proactive surveys for sensitive and protected/regulated species, (3) monitoring long-term population trends, and (4) collaborating with other agencies.

Preactivity surveys are conducted in areas where surface-disturbing work will occur as part of a project. These surveys are conducted by biologists who search the project area for sensitive or protected/regulated species or their sign. Historic data is also referenced to see if there are known sensitive or protected/regulated species in the

project area. Pre-activity surveys are specifically designed to protect sensitive and protected/regulated species from being harmed. If a project occurs within desert tortoise habitat, a desert tortoise clearance survey is conducted in addition to the preactivity survey (see desert tortoise section below).

Proactive surveys are conducted for sensitive and protected/regulated species in order to define their distribution on the NTS and learn more about their natural history including identifying important habitat features that are necessary for their survival (for example, bat roosts, nesting sites). Proactively monitoring sensitive and protected/regulated species can provide data that limits mitigation measures to the known distribution of a species in the event it becomes listed (for example, desert tortoise) or prevents the listing of certain species such as Beatley milkvetch (*Astragalus beatleyae*). Historic data collected by other researchers (for example, Brigham Young University, Nevada Applied Ecology Group, International Biome Program, Basic Environmental Compliance and Monitoring Program) has proved to be invaluable and a great foundation on which to build. Opportunistic sighting data is also used to help understand distribution patterns, although it is understood that there are limitations to these data because they are not collected systematically and uniformly across the NTS. Most of the historic data and recently collected data have been entered into electronic databases that can be linked to a geographic information system to produce distribution maps and delineate geographic areas or habitat types where sampling is incomplete.

Monitoring long-term population trends in wildlife and their habitat is necessary to determine if NTS activities are impacting these resources. However, because natural variability is high in NTS ecosystems due to limited and unpredictable moisture it can be difficult to determine if a downward population trend is caused by NTS activities or is just due to natural variability. Results from long-term monitoring have helped biologists understand some of these patterns but ongoing monitoring is still needed. Learning from other researchers and being familiar with the historic and current literature is also a big part of monitoring population trends. New techniques and statistical analyses for monitoring wildlife populations are continually being developed. Some of these (for example, program PRESENCE, MacKenzie and others [2002]) are being evaluated to see if they are applicable for monitoring certain species on the NTS. Long-term habitat monitoring helps to understand natural changes in the vegetation (for example, substantial decline in spiny hopsage (*Grayia spinosa*) and the impacts of revegetation and succession on wildlife after natural or human-caused disturbance.

Collaborating with other agencies and biologists is critical to wildlife monitoring success on the NTS due to limited funding, a reduced work force, and lack of expertise in certain areas. Many agencies and institutions face similar challenges, and working together creates a win-win situation for everyone. Attending and presenting at professional meetings and involvement with professional societies is a great way to meet and collaborate with other biologists. Additionally, having a good working relationship with regulatory agencies such as FWS and state wildlife agencies can result in expedited consultations and facilitates the issuing of permits. Another way we collaborate with other agencies is to produce an annual report that summarizes our work for each calendar year as well as topical reports on specific studies that have been completed. These reports are distributed to other land management agencies in Nevada and are available to the public through the Office of Scientific and Technical Information (OSTI). The following sections contain information about ways NSTec biologists have made monitoring more efficient and detail examples of successful monitoring and collaborations.

Fairy Shrimp

Fairy shrimp are unique organisms that only hatch under the right conditions (for example, after significant precipitation events). Eggs may lay dormant in sediments and can be viable for years. They are not sensitive or protected/regulated but more needs to be done to determine their temporal and spatial distribution. Surveys were initiated to determine what species were present and where they occurred. Specimens were sent to Christopher Rogers with EcoAnalysts Incorporated for identification and verification. Mr. Rogers is a fairy shrimp expert and agreed to collaborate with us. Based on his preliminary analysis, it is possible that a new species of the genus *Branchinecta* has been discovered, but further work is needed to verify the new species.

West Nile Virus

West Nile Virus (WNV), a potentially deadly disease transmitted through mosquito bites, was first detected in southern Nevada during the spring of 2004. During the summer of 2004, Southern Nevada Health District (SNHD) personnel came to the NTS and trained NSTec biologists to trap mosquitoes, and they also provided all the trapping equipment. NSTec biologists have continued trapping mosquitoes every year since 2004. Mosquitoes collected from trapping are given to SNHD personnel who identify the mosquitoes and test them to see if they have WNV. Results are used to evaluate the risk of NTS workers contracting WNV, and SNHD includes the NTS data in their reports to document the presence of WNV throughout southern Nevada. Results are also used to document mosquito distribution on the NTS which has not been done before.

Western Red-tailed Skink

The western red-tailed skink (*Eumeces gilberti rubricaudatus*) is a sensitive species that was historically known from four locations on the NTS. Monitoring was initiated in 2006 to define its distribution and determine the best technique for capturing them. In 2007 and 2008, systematic sampling across the NTS was conducted. There are now 13 known skink locations on the NTS. In the process of trapping for western red-tailed skinks, other reptile species are also captured. This greatly expands the known distribution of several species, especially since trapping is conducted in many parts of the NTS and in habitats that have never been sampled before. One example of this is the capture of three Great Basin skinks (*Eumeces skiltonianus*) on Pahute Mesa where it was previously only known to occur on Rainier Mesa. Additionally, in order to maximize our time in the field small mammal trapping is conducted in conjunction with reptile trapping activities. This greatly reduces the time and effort required to document species distributions, especially since many trapping locations are in rugged, remote areas. Opportunistic sighting records of other species such as birds or large mammals are also documented during reptile trapping to better define species distributions.

Additionally, we are collaborating with Dr. Jonathan Richmond, a skink expert at Cornell University. This has benefited both Dr. Richmond and NSTec biologists by providing him with tissue samples of both western red-tailed and Great Basin skinks for genetic testing and NSTec biologists with the results of the genetic testing and the potential for future identification of family groups and dispersal in NTS populations. He is discovering new information about the evolutionary ecology and phylogenetic relationships of these two skink species and filling in data gaps where no data had been collected before. Dr. Richmond also visited the NTS in May 2008, taught NSTec biologists new techniques for capturing skinks and helped identify locations to sample for skinks, and was able to expand his knowledge of western red-tailed skink habitat.

Desert Tortoise

The desert tortoise is listed as Threatened under the Endangered Species Act. Therefore, NTS activities must comply with the terms and conditions in a FWS Biological Opinion. For instance, before any surface-disturbing work occurs in tortoise habitat, a desert tortoise clearance survey is conducted in which 100 percent of the project area and a buffer area is surveyed to determine if tortoises occur in the project area. If a tortoise is present and the project cannot be postponed or moved, the tortoise is moved to a new burrow a safe distance away from the project area. The presence of other sensitive and protected/regulated species is also documented, and mitigation actions are taken to protect these as well.

The desert tortoise monitoring program is a good example of how proactive monitoring can save a lot of time and money. Surveys were conducted to determine the distribution of the desert tortoise before it became listed as "Threatened" under the Endangered Species Act. Thus, when it became listed, the previously gathered data was used in consultation with FWS and the mitigation actions and terms and conditions of the Biological Opinion were limited to the known tortoise distribution.

Western Burrowing Owl

The western burrowing owl is protected under the Migratory Bird Treaty Act, and is a FWS National Species of Conservation Concern because it is declining in many parts of its range. We monitored this species for several years and produced a topical report entitled "Ecology of the Western Burrowing Owl on the Nevada Test Site." (Bechtel Nevada 2003). It covered several aspects of western burrowing owl natural history and ecology including information about the distribution of this species on the NTS, burrow occupancy patterns based on monthly burrow monitoring, reproduction (in other words, number of breeding pairs and number of young per breeding pair) using motion-activated cameras, food habits based on pellet analysis, disturbance effects, and burrow temperatures using data loggers.

The last few years we have collaborated with Dr. Courtney Conway (University of Arizona) who is conducting a study on migratory patterns and relatedness among western burrowing owls throughout the western United States. He provides the training, the necessary permits, and all the supplies and we provide the labor. This collaboration benefits us by providing data on philopatry and potential wintering sites for NTS owls, and he benefits by getting data that would be difficult and costly to obtain.

Mule Deer

Mule deer are protected and regulated by the state of Nevada as a game species, although hunting has not been allowed on the NTS since the 1940s. In order to document long-term population trends based on relative abundance, spotlight surveys have been conducted periodically since 1989 (NSTec 2008). Survey routes were designed based partly on a study by Giles and Cooper (1985) who used radiotelemetry to determine distribution patterns of mule deer on the NTS. Based on survey counts, the population of mule deer appears to have declined from 1989 to 1994 reaching its lowest point in 1999 and 2000. In 2006, the population was greatest and then declined by half in 2007. Drought, water source availability, predation, and habitat modification from wildfire appear to have had the greatest influence on relative abundance. Knowing the long-term population trends of mule deer is instructive and may be correlated with puma numbers.

Puma

Pumas are also protected and regulated by the state of Nevada as a game species. No formal study has been conducted on pumas on the NTS before. In fact, very few puma studies have been conducted in Nevada. We have 87 opportunistic records of puma sightings or sign on the NTS, and over the last few years, several records of puma presence around active projects and facilities have been documented. Since 2005, we have collaborated with Dr. Erin Boydston, a United States Geological Survey research scientist, who purchased several motion-activated cameras to investigate puma abundance and distribution on the NTS. Almost 60 photographs of pumas have been taken at six sites with as many as five to ten individuals occurring at one site. There appears to be a healthy, reproductive population of pumas in the northern portion of the NTS as evidenced by pictures of mature males, mature females, lactating females, and subadults. We are currently in the process of collaborating with Nevada Division of Wildlife and University of Nevada, Reno to conduct a radiotelemetry study to track puma movements and identify how much time they are spending around active projects and facilities to assess potential risk to NTS workers and learn more about puma natural history and ecology. The NTS is an ideal place to study pumas because they have not been hunted for several decades so data from this area can be compared to other areas where hunting is allowed. This is one reason why Nevada Division of Wildlife is very interested in collaborating with us.

In conclusion, we have found that a major key to successful wildlife monitoring is collaboration with other agencies and biologists. Sharing resources and knowledge saves time and money and usually results in better studies. Another key to success is to monitor multiple species when possible to make the most of your time in the field. Proactive monitoring of sensitive and protected/regulated species is also very important and can result in reducing mitigation impacts and even prevent the listing of some species.

Other aspects of successful monitoring include building on existing or historic data, knowing the literature pertaining to

the species you are monitoring, and being involved with professional organizations. These allow you to learn from other people's mistakes and successes and help avoid unnecessary repetition.

ACKNOWLEDGMENTS

The authors would like to thank Dennis J. Hansen and Sierra Cory for helpful reviews of this paper. This manuscript has been authored by National Security Technologies, LLC, under Contract No. DE-AC52-06NA25946 with the U.S. Department of Energy. The United States Government retains and the publisher, by accepting the article for publication, acknowledges that the United States Government retains a non-exclusive, paid-up, irrevocable, world-wide license to publish or reproduce the published form of this manuscript, or allow others to do so, for United States Government purposes.

REFERENCES

- Bechtel Nevada. 2003. Ecology of the western burrowing owl on the Nevada Test Site. DOE/NV/11718--701, Las Vegas, 73 p.
- DOE/NV. 1996. Final Environmental Impact Statement for the Nevada Test Site and Off-site Locations in the state of Nevada. Volume 1, Chapters 1-9. DOE/EIS 0243. U.S. Department of Energy, Nevada Operations Office, Las Vegas, NV.
- DOE/NV. 1998. Nevada Test Site Resource Management Plan. DOE/NV--518, Las Vegas, NV.
- Giles, K.R., Cooper, J. 1985. Characteristics and migration patterns of mule deer on the Nevada Test Site. DOE/DP/00539-054 Environmental Monitoring Systems Laboratory, U.S. Environmental Protection Agency, Las Vegas, NV. 17 p.
- Jaeger, E.C. 1957. The North American Deserts. Stanford: Stanford University Press. 308 p.
- MacKenzie, D.I., Nichols, J.D., Lachman, G.B., Droege, S., Royle, J.A., Langtimm, C.A. 2002. Estimating site occupancy rates when detection probabilities are less than one. *Ecology* 83(8):2248-2255.
- NSTec. 2008. Ecological Monitoring and Compliance Program 2007 Report. DOE/NV/25946--402, Las Vegas, NV. 125 p.
- Ostler, W.K., Hansen, D.J., Anderson, D.C., Hall, D.B. 2000. Classification of vegetation on the Nevada Test Site. DOE/NV/11718--477. Bechtel Nevada, Las Vegas, NV.
- Western Bat Working Group. 1998. Western Bat Species: Regional Priority Matrix. Reno, NV. 4 p.
- Wills, C.A., Ostler, W.K. 2001. Ecology of the Nevada Test Site: an annotated bibliography. DOE/NV/11718--594. Bechtel Nevada, Las Vegas, NV. 386 p.

Long-Term Changes in Southern Utah Upland Shrub Communities: A Study in Repeat Photography

Charles E. Kay¹ and Chad R. Reid²

ABSTRACT

To evaluate long-term range conditions in southern Utah, 1,879 repeat photographs were made on the Dixie and Fishlake National Forests, and adjoining lower elevation BLM and private lands. Repeat photographs were also made in Zion National Park, Cedar Breaks National Monument, and Bryce Canyon National Park where livestock grazing has been prohibited for many years. Of the total, 926 photo-pairs depict sagebrush (*Artemisia* spp.) communities and 471 depict mountain brush associations (*Cercocarpus ledifolius*, *C. montanus*, *Amelanchier utahensis*, *Quercus gambelii*, *Prunus virginiana*). Sagebrush decreased in 261 photosets, increased in 386 and remained unchanged in 279, while mountain brush decreased in 10 photopairs, increased in 440, and showed no change in 21. Sagebrush increased on all sites that had been treated and planted to crested wheatgrass (*Agropyron desertorum*) during the 1950s to 1960s. Most of the decline in sagebrush was due to invasion by pinyon (*Pinus* spp.) and/or juniper (*Juniperus* spp.), while an absence of fire is the most likely reason mountain brush increased. The observed changes have major implications for wildlife, and especially mule deer (*Odocoileus hemionus*) where sagebrush winter ranges have been lost or where mountain brush has grown beyond the reach of browsing animals. All repeat-photosets and descriptive text have been placed on Utah State University's Extension website and can be accessed at <http://extension.usu.edu.rra>.

INTRODUCTION

Certain segments of the public have expressed concern that livestock may be degrading Utah's rangelands, while others fear that logging is destroying Utah's forests. According to some, mineral exploration, including oil and gas activity, is permanently scarring Utah's environment. In addition, governmental agencies are starting to manage for a more "natural" landscape, but what were the original conditions in Utah? And are our rangelands being overgrazed? Are Utah's riparian areas being ravaged? Our forests denuded? Our lands scarred?

One way to address these and similar issues is through repeat photography, where scenes depicted in historical photographs are rephotographed as they appear today, forming sets of images taken or repeated, from the same camera stations (Hall 2002a, 2002b; Magill 1989; Rogers and others 1984). This works best for vegetation types

that are clearly identifiable by photographic analysis, such as grasslands, sagebrush (*Artemisia* spp.), pinyon-juniper (*Pinus* spp.-*Juniperus* spp.), aspen (*Populus tremuloides*), conifers (*Abies* spp., *Picea* spp., *Pinus* spp., and *Pseudotsuga* spp.), and willows (*Salix* spp.). Repeat photographs can also be used to estimate rates of soil erosion and plant cover (Rogers and others 1984).

Repeat photographs are extremely valuable because they provide a long-term perspective that is often missing from other studies and because the photo-pairs can readily be interpreted by the public. Want to know what things were like 100 year ago? Find an old photograph. Want to see how things have changed? Locate the original camera station and rephotograph the scene as it appears today. While the process seems simple, it can often be very time consuming because early photographers often failed to record exactly where their pictures were taken. Sometimes it can take days to find the camera station for a single photograph.

The present repeat-photo project was initiated in southwestern Utah at the request of Utah State University Extension Range Specialists and Forest Service officials, who were concerned about the long-term vegetation changes they suspected were occurring on the Fishlake and Dixie National Forests and surrounding areas. Various studies were begun to quantify the range of historical variability, but it was thought that repeat photographs might best convey past and present conditions to the general public. After all, if a picture is worth a thousand words, repeat-photographs are priceless (Kay 2006).

This paper reports on changes that have occurred in sagebrush and mountain brush (*Cercocarpus ledifolius*, *C. montanus*, *Amelanchier utahensis*, *Quercus gambelii*, and *Prunus virginiana*; among others) communities. Papers on aspen, pinyon-juniper, conifers, woody riparian vegetation, and general range conditions will be published elsewhere. It must also be remembered that Europeans began to settle Utah in 1847. Thus, even the earliest photographs often reflect years of livestock grazing, except for the 1872 photographs taken on Boulder Mountain by John Hillers as part of the Powell Survey (Fowler 1989) since that portion of Utah was not settled by Europeans or grazed by livestock until after 1875 (LeFevre 1973; Woolsey 1964). To the best of our knowledge, this is one of only five sets of early photographs that depict vegetation conditions prior to the onset livestock grazing in western North America - a rare ecological benchmark.

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Charles E. Kay, Department of Political Science, Utah State University, Logan, UT

²Chad R. Reid, Utah State University Extension, Cedar City, UT

METHODS

We first searched the following archives for historical photographs taken in southwestern Utah: (1) The U.S. Geological Survey's photographic library in Denver, CO; (2) the Forest Service's Regional Office photographic collection housed at Weber State University, Ogden UT; (3) the Utah Historical Society in Salt Lake City, UT; (4) LDS Church Historical Department Archives in Salt Lake City, UT; (5) Southern Utah State University's Archives and Special Collections, Cedar City, UT; (6) University of Utah's Archives and Special Collections, Salt Lake City, UT; (7) the historical files kept by the Supervisor's Office on the Dixie National Forest in Cedar City, UT; (8) the U.S. Geological Survey repeat-photo collection house at Arizona State University, Tucson, AZ; (9) Bryce Canyon National Park photo archives, Bryce Canyon, UT; (10) Zion National Park photo archives, Springdale, UT; (11) the range files held by the Panguitch Ranger District, Dixie National Forest, Panguitch, UT; (12) the range files held by the Teasdale Ranger District, Dixie National Forest, Teasdale, UT; (13) the range files held by the Escalante Ranger District, Dixie National Forest, Escalante, UT; (14) the range files held by the Cedar City Ranger District, Cedar City, UT; (15) the ranger files held by the Pine Valley Ranger District, St. George, UT; (16) Allen DeMill's private collection, Kanab, UT; (17) Worth Brown's private collection, Kanab, UT; (18) the Natural Resources Conservation Service's Snow Survey records, Salt Lake City, UT; (19) Utah State University's Archives and Special Collections, Logan, UT; (20) Kanab City Museum, Kanab, UT; (21) the National Archives, Washington, D.C.; and (22) Lynne Clark's private collection, St. George, UT.

Photo Plot Transects

During the early 1940s, a series of Photo Plot Transects were established on U.S. Forest Service lands throughout Utah to monitor range conditions. The variable-length transects consist of a linear line with three foot square plots (9ft²) at 100 ft, 200 ft, or longer intervals. Basal area was recorded for every plant species on each subplot and close-up photographs were taken each time the subplots were visited. General landscape photos were also taken each time the transects were measured. The Photo Plot Transects were all read twice during the 1940s, twice during the 1950s, and once in 1979. As the Photo Plot Transects were initially sited on key areas located throughout each Forest District, they are an invaluable barometer of long-term range conditions. The existence of these transects was first brought to our attention in 2003 and over the last five years, all Photo Plot Transects on the Dixie National Forest were revisited. The general landscape photographs associated with each transects were retaken. Photo Plot Transects on the Fishlake National Forest were not visited, as we were not made aware of those photos until after the Fishlake part of this project was completed. The same is true for Parker Three-Step Transects.

Parker Three-Step Transects

During the 1950s and 1960s, the Forest Service established a number of Parker Three-Step Transects to monitor range conditions on many grazing allotments. Each transect consisted of one, two, or in some cases three, 100 ft lines designed to measure plant species composition. In addition, landscape photographs were taken of each line and transect. Although the Parker Three-Step method is no longer utilized by land management agencies (Cook and others 1992; Reppert and Francis 1973), the associated photographs can be used to evaluate long-term range conditions. Thus, all range files on the Teasdale, Escalante, Panguitch, Cedar City, and St. George Ranger Districts on the Dixie National Forest were searched for Parker, as well as other old photographs.

As all Photo Plot Transects and Parker Three-Step Transects were marked with steel stakes, which for the most part still exist, and because Forest Service files usually contain hand drawn site maps, we were able to repeat most of the associated landscape photographs. These, along with historical images obtained from other sources, produced a complete coverage of the vegetation types on the Dixie National Forest, - - - something that has seldom been accomplished in other repeat-photo studies. Thus, we believe this is the most detailed repeat-photo project that has ever been undertaken for a comparable area anywhere in the western United States.

Photography

Repeat photographs were taken with both color slide film (Fujichrome 100) and black and white print film (Kodak Plus-X). We used a Nikon FM2 with a 24 to 50mm lens and a L-37 filter for the color slides, and a Nikon FM2 with a 24 to 50mm lens and a Y-48 filter for the black and white prints. Nikon 28 to 80mm and 80 to 200mm lens were also used, as needed, to match original focal lengths. Multiple images were made at each site with both color slide and black and white print film. Depending on the focal length of the original photograph, wider angle retakes were also made. All cameras were hand-held and retakes were generally made at the same time of day as the original photographs. In addition, each photopoint was marked on U.S. Geological Survey 1:24,000 topographic maps. Camera station elevation and UTM coordinates were acquired either via GPS or estimated from topographic maps. While at each site, a detailed description of the present vegetation was recorded, as were any changes from the original photograph. Upon returning from the field, a systematic evaluation of all the photosets was conducted and detailed descriptions written for each photopair (Kay 2003). Each photoset was visually evaluated and vegetation types were recorded as having increased, remained the same, or decreased in aerial extent, height, and density (Kay 2003). The photosets or Plates were consecutively

numbered according to the order in which the repeat photographs were completed. Plates on the Fishlake were numbered from 1 to 355, while those on the Dixie were numbered from 1x to 1524x.

RESULTS AND DISCUSSION

To eliminate possible bias, we attempted to locate and repeat all the early photographs that existed in major collections for the Fishlake and Dixie National Forests and surrounding areas. To be repeated, however, photographs need to contain features that could be relocated in the field. Close-ups of vegetation, or pictures in which there were no clearly identifiable landmarks, proved impossible to relocate, despite help from long-time area residents. In all we were able to complete 1,879 repeat photographs and as those photosets cover a broad array of vegetation types, time periods, elevation zones, and landforms, (figures 1 and 2), we believe they are representative of the changes that have occurred throughout southern Utah.

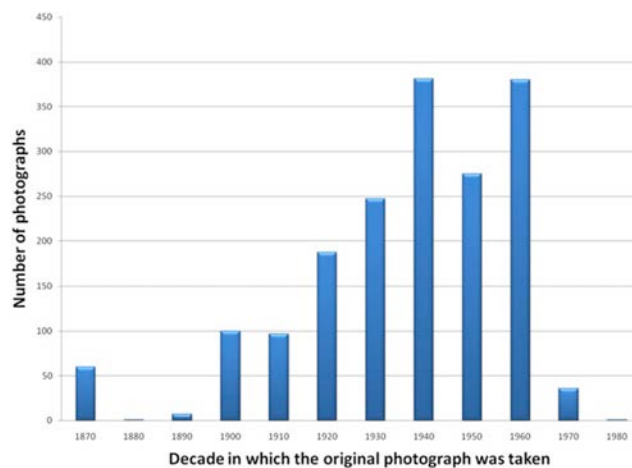


Figure 1—The decade in which the original photographs were taken on the Fishlake and Dixie National Forests ($n = 1,879$).

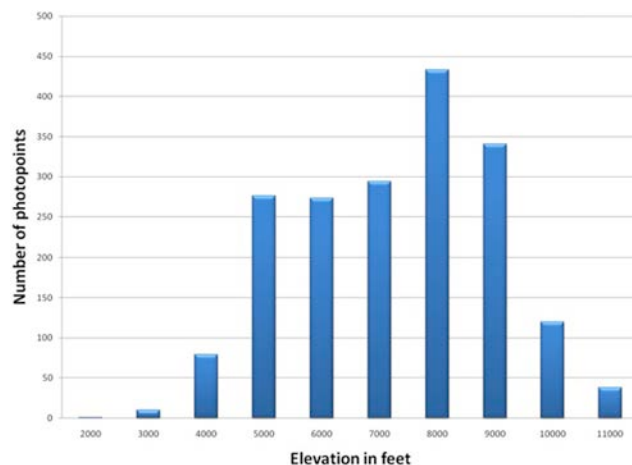


Figure 2—Photopoint elevation on the Fishlake and Dixie National Forests ($n = 1,879$).

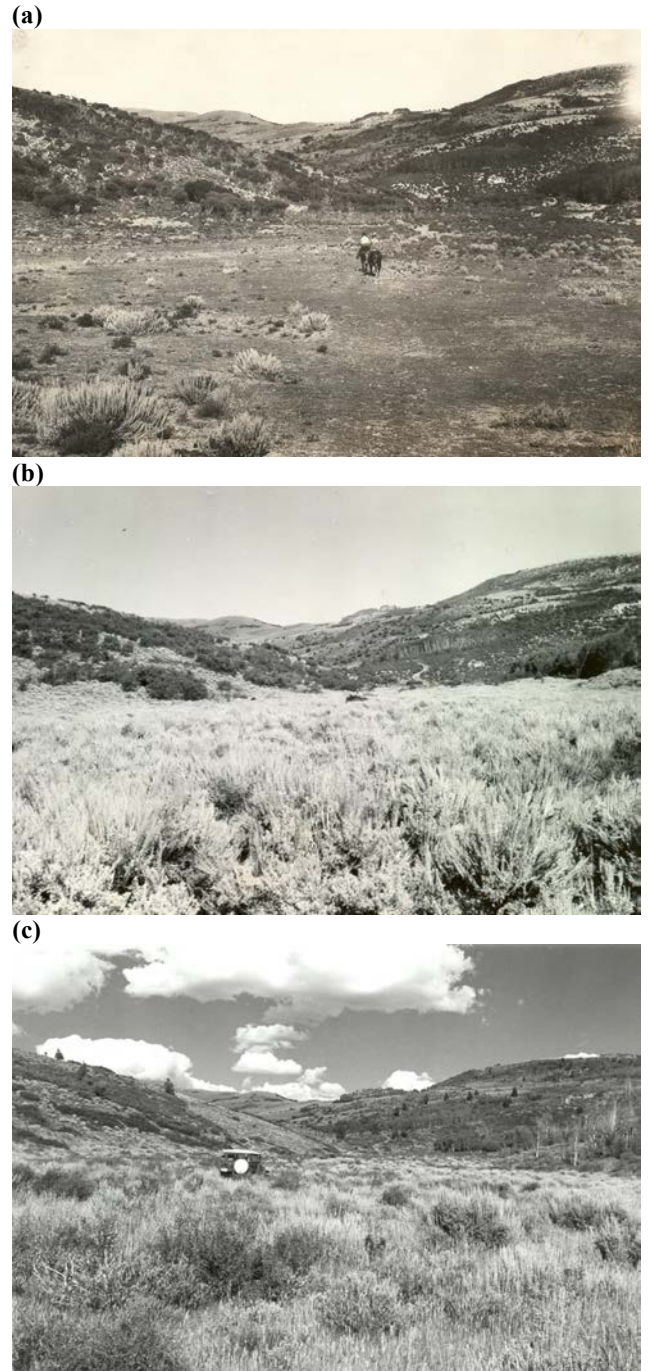


Figure 3—Sagebrush increased in more than 40 percent of the Fishlake and Dixie repeat photosets. (a) The range was very heavily grazed on Sargent Mountain in 1919 - - note the horsemen for scale (Forest Service photo 41735A). (b) The Forest Service instituted measures to reduce grazing and by 1943 sagebrush had increased - - again note the horseman in the distance for scale (Forest Service photo 468731). (c) Today, though, sagebrush cover has declined as the site has been prescribed burned by the Forest Service (Charles E. Kay photo 4446-13). For more details see Plate 46 on USU Extension's website.

Of the 1,879 total, 926 photo-pairs depict sagebrush communities and 471 depict mountain brush associations. Sagebrush decreased in 261 photosets, increased in 386, and remained unchanged in 279, while mountain brush decreased in 10 photo-pairs, increased in 440, and showed no change in 21. That is to say, sagebrush increased in 42 percent of the photosets, declined in 28 percent and remained about the same in 30 percent. While mountain brush increased in 93 percent of the repeat photographs, declined in 2 percent and showed no change in 5 percent. All repeat photographs and descriptive text have been placed on Utah State University's Extension website and can be accessed at <http://extension.usu.edu/rra>. This is the only large-scale, repeat-photo project where all the completed photosets and supporting information are available through the internet to land managers and the general public. As only a small selection of photo-pairs can be included in this publication, we encourage readers to view the entire collection on USU Extension's website.

Sagebrush

Sagebrush increased in density, cover, and aerial extent on most sites that had been heavily grazed during the late 1800s and early 1900s (figure 3). This is similar to what others have reported in Utah and across the Intermountain West (Christian and Johnson 1964; Cottam 1947; Cottam and Stewart 1940; Hall and Cottam 1955; Stewart 1941). Sagebrush declined on a few sites due to wildfire, agricultural clearing, or industrial development — for example, see Plates 179, 479x, and 1474x on USU Extension's website. However, conifer encroachment was by far and away the number one reason sagebrush has declined in southern Utah. Many sagebrush stands were lost to ever expanding pinyon-juniper forests (figure 4), which is not surprising because pinyon-juniper increased in 96 percent of our repeat photographs in which it occurred. Again this is similar to what has been reported in Utah and throughout the western United States (Burkhardt and Tisdale 1976; Ffolliot and Gottfried 2002; Johnson and Miller 2008; Miller and Rose 1995; Miller and Wigard 1994; Miller and others 2008; Yorks and other 1994).

Treated Sagebrush

During the 1950s, 1960s, and early 1970s, the Forest Service and other land management agencies attempted to improve range conditions by using herbicides or mechanical treatments to reduce the cover of sagebrush prior to reseeding with crested wheatgrass (*Agropyron desertorum*) or other non-native species. Concerns have now been raised that this created millions of ac of monoculture throughout the West. Repeat photographs, though, indicate that this has not happened in southern Utah.

Of the 14 photo-pairs on the Dixie and Fishlake that depict sprayed sagebrush stands, all show that sagebrush has returned to its former abundance or actually increased. For

instance, see Plates 213, 4x, 5x, 197x, 198x, and 199x on USU Extension's website. Of the 99 photosets that depict root-plowed and reseeded sagebrush stands, 94 percent show that sagebrush has now reinvaded those sites (figures 5 and 6). Others have reported similar trends on treated sagebrush sites across the West (Johnson 1969; Thilenius and Brown 1974; Wambolt and Payne 1987; Watts and Wambolt 1996).

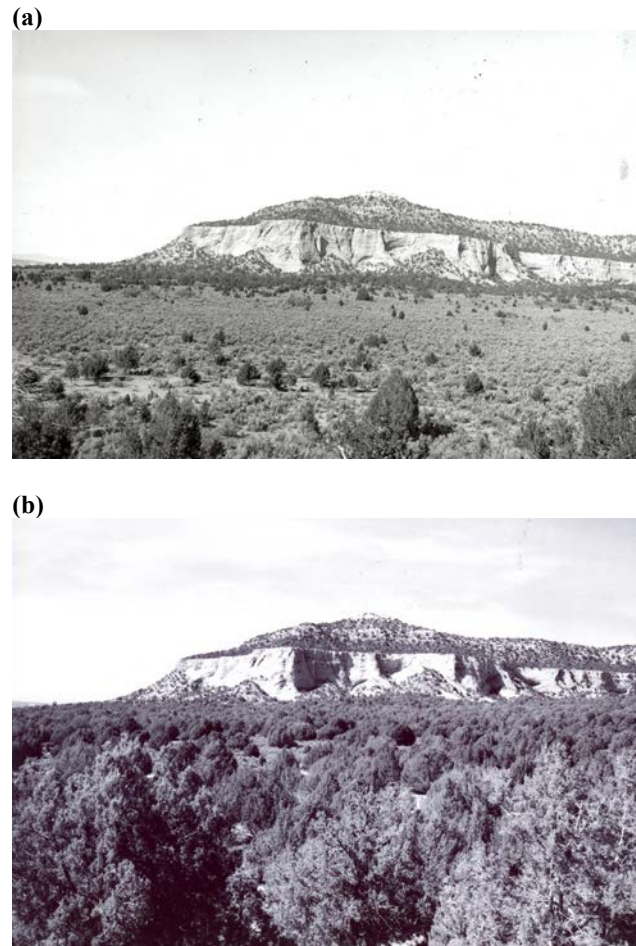


Figure 4—Most of the decline in sagebrush on the Fishlake and Dixie National Forests was due to expanding pinyon-juniper woodlands. (a) Pinyon and juniper had started to invade the sagebrush flat below Harris Point in 1936 and by (b) 2006 there was very little sagebrush left. 1936 photo by the U.S. Geological Survey (J.C. Anderson 5) and 2006 photograph by Charles E. Kay (photo 5633-6). For additional details see Plate 1190x on USU Extension's website.

Mountain Brush

On most sites in southern Utah, mountain brush species have increased in height, density, and aerial extent (figure 7). This is most likely due to an absence of fire and a reduction in livestock grazing (Austin and others 1986). On a few sites, however, ponderosa pine (*Pinus ponderosa*) and white fir (*Abies concolor*) have begun to invade mountain brush communities (figure 8). This is not surprising since

92 percent of the photosets that depict upland forests on the Fishlake and Dixie show that conifers have increased, often markedly.

Other Shrubs

On some higher-elevation rangeland sites, shrubs such as snowberry (*Symphoricarpos oreophilus*) and currants (*Ribes* spp.) have increased (figure 9). This is most likely due to the elimination of grazing by domestic sheep, as many Forest Service allotments have been converted from sheep to cattle over the years. Cattle do not normally graze either snowberry or currants, but sheep will, especially on heavily stocked ranges.

Why Have Things Changed?

Other repeat-photo studies have reported the same long-term vegetation changes that we documented in southern Utah (Bradford and others 2004; Creque and other 1999; Rogers 1982; Turner and other 2003; U.S Forest Service 1993; among others). Several hypotheses have been proposed to explain why woody vegetation has increased on western rangelands, as well as on rangelands around the world (Branson 1985; Britz and Ward 2007; Pyne 1991; Roques and others 2001; Van Auken 2000; among others). These include climatic change, excessive livestock grazing, increase in atmospheric carbon dioxide, and fire suppression by land management agencies.

Sagebrush and other unpalatable woody species do increase with heavy livestock use (Young 1989) but pinyon and juniper have also invaded ungrazed sagebrush communities (Knapp and Soule 1998; Soule and Knapp 1999, 2000). Although sagebrush and other shrubs often respond positively to increased moisture, climate alone has not been shown to be the main factor in brush encroachment. Similarly, while woody species grow better in a carbon dioxide enriched environment, carbon dioxide alone has not caused the increase in woody vegetation (Archer and other 1995; Johnson and others 1993; Knapp and Soule 1996), which leaves fire, or the lack thereof, as the most likely reason woody plants have increased in Utah and throughout the West (Bond and Keeley 2005; Bond and others 2005; Lesica and others 2007; Miller and Tausch 2002).

While we agree that the lack of fire is the most parsimonious explanation as to why woody vegetation has increased on western ranges, we do not believe the cause lives solely with fire suppression. Vale (2002) and others contend that lightning historically caused most range and forest fires, but that appears not to be true. Kay (2007) recently compared known lightning-ignition rates in the United States with potential aboriginal-ignition rates, based on pre-European contact estimates of native populations and estimates of how many fires each native person set per year.

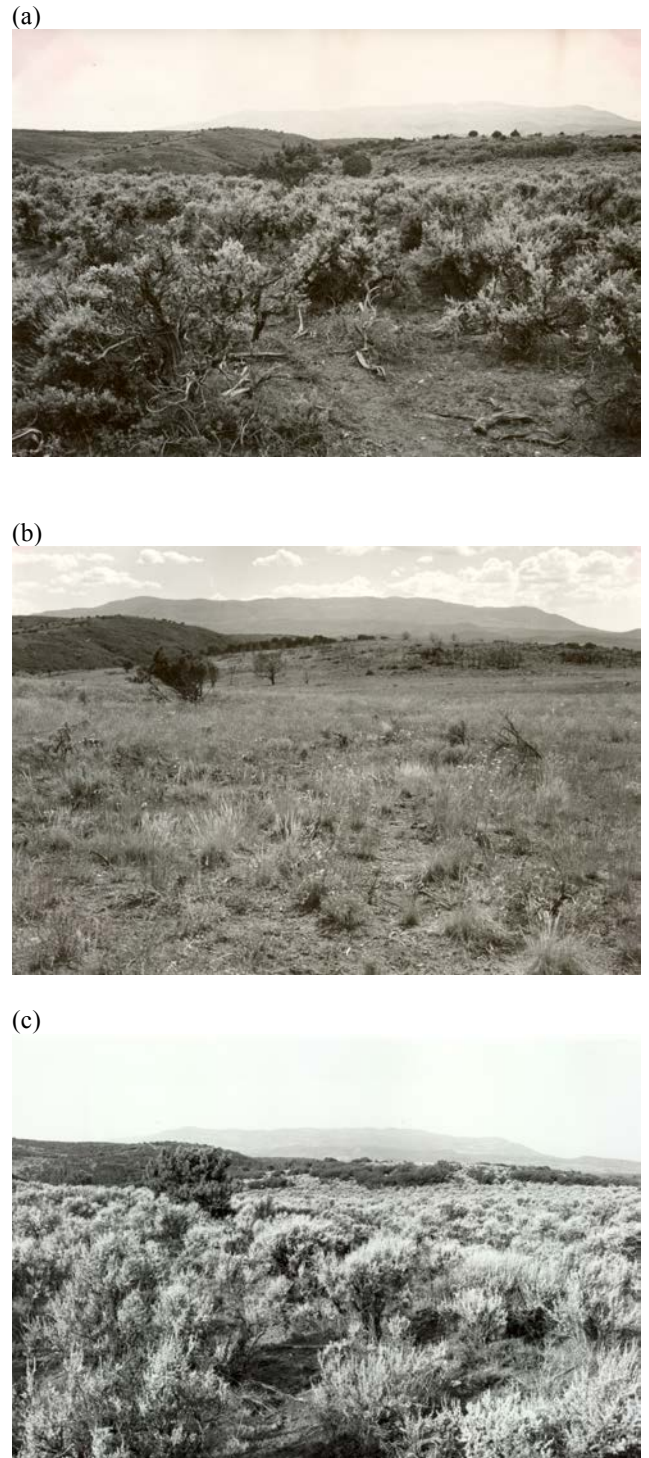


Figure 5—A sagebrush stand on the Fishlake National Forest that was root-plowed, burned, and reseeded. (a) The site as it looked before treatment in 1949. (b) The same site in 1954 after it had been root-plowed, burned, and reseeded to crested wheatgrass. (c) By 2000, however, sagebrush again dominated the area. Photos (a) and (b) by the Forest Service (unnumbered), 2000 photo by Charles E. Kay (4651-14). For more details see Plate 220 on USU Extension's website.

“Using the lowest published estimate of native people in the United States and Canada prior to European influence (2 million) and assuming that each individual started only 1 fire per year -- potential aboriginal ignition rates were 2.7 to 350 times greater than current lightning ignition rates. Using more realistic estimates of native populations, as well as the number of fires each person started per year, potential aboriginal ignition rates were 270 to 35,000 times greater than known lightning ignition rates. Thus, lightning-caused fires may have been largely irrelevant for at least the last 10,000 years. Instead, the dominant ecological force likely has been aboriginal burning” (Kay 2007:16).

In the Great Basin, for instance, even using a low estimate of only one native person per 9 mi² (Vale 2002:81) and assuming that only one fire was set per person per year, native people would still have started 10 times more fires than lightning (Kay 2007:20). Using more realistic estimates of the number of native people in the Great Basin prior to European influences, such as smallpox which decimated native populations throughout North America ca. 1550, aboriginal-started fires were likely two or three orders of magnitude greater than known lightning-ignition rates. As discussed elsewhere, native people made extensive use of fire to manipulate their environment and to make it more productive (Anderson 2005; Kay 2007; and references therein). In fact, native people actually created many ecosystems heretofore thought to be “natural” (Kay 1995, 2003, 2007).

Impact on Wildlife

As with any vegetation change, some animals will benefit, while others will not. Two species that are harmed by the loss of sagebrush are sage grouse (*Centrocercus urophasianus* and *C. minimus*) and mule deer (*Odocoileus hemionus*) (Crawford and others 2004; Kay 2006). As pinyon and juniper invade critical sagebrush winter ranges (figure 10), mule deer populations decline, as there are fewer shrubs for the wintering deer to eat. In fact, forage loss due to increasing woody vegetation has been identified as a major problem on western ranges (Kay 2003:18-19). While fire certainly reduces the cover of sagebrush at least temporarily, it is still one of the best ways to rejuvenate decadent sagebrush communities and to create the diversity of different age sage, as well as forbs, needed by sage grouse and other sage obligate species (Crawford and others 2004).

Other Considerations

The response to our repeat-photo work in southern Utah has been extremely positive and we have made invited presentations to various user groups. As all our photosets are available over the internet, they have been widely used by the public, as well as land managers. Based on our experience, we believe that every National Forest, BLM Resource Management Area, National Park, National Wildlife Refuge, and the like should undertake their own repeat-photo projects. Data on the range of historical

variability are fine, but the general public is not trained to make sense of oftentimes complicated datasets. Moreover, many segments of the public do not trust the agencies to collect unbiased data. On the other hand, just about everyone can look at an old photograph and a modern retake and see for themselves what has changed. Besides, given our modern visual culture, people tend to believe what they see with their own eyes. This is one reason why digital photography should never be used in repeat-photo work -- because the images are too easy to manipulate. In addition, all repeat-photo work should include black and white photography. This is because black and white prints and negatives, if properly stored, will last forever while color photographs start to deteriorate as soon as they are made, even if stored under archival conditions.

(a)



(b)



Figure 6—A sagebrush stand that was treated on the Dixie National Forest. The site is on Sheep Creek and the view is west to Bryce Canyon National Park. (a) The area as it looked in 1961 after it had been root-plowed to remove sagebrush and reseeded to crested wheatgrass. (b) That same site 43 years later in 2004. Sagebrush has reinvaded the area and pinyon-juniper encroachment has begun. 1961 photograph by the Forest Service (unnumbered), while the 2004 retake was made by Charles E. Kay (5287-12). For additional information see Plate 401x on USU Extension’s website.

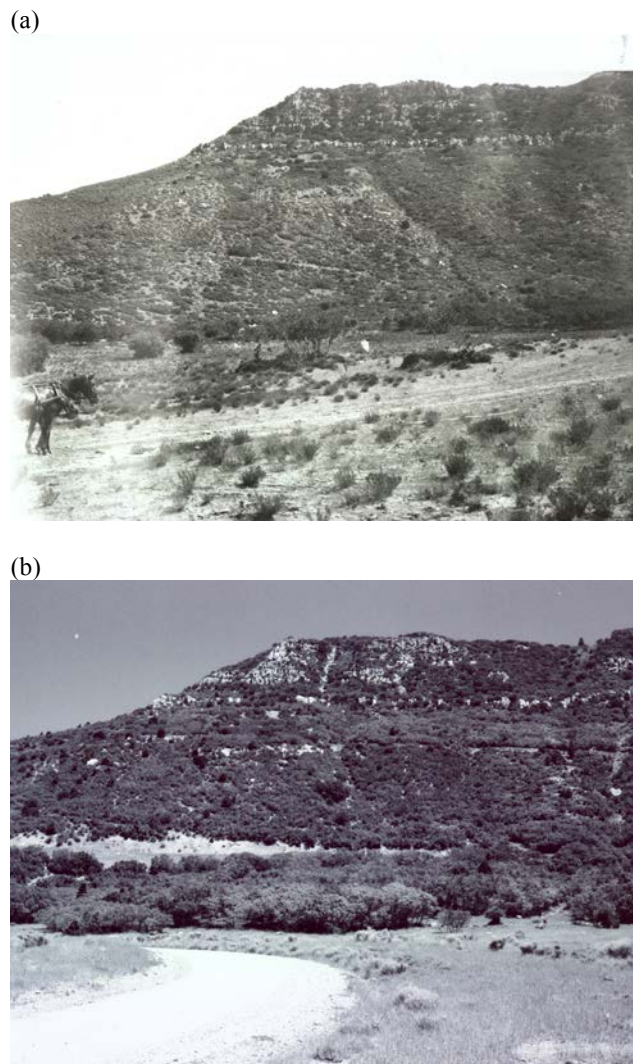


Figure 7—100 years of vegetation change in the mountain brush zone on Lone Tree Mountain southeast of Cedar City, Utah. (a) In the 1906 image, the shrubs are low-statured and widely spaced. The high-lined shrubs in the foreground indicate the area was heavily grazed, most likely by domestic sheep. U.S. Geological Survey photograph (W.T. Lee 219). (b) Today, the shrubs have increased in height, cover, and probably density, as well. Invasion by pinyon-juniper has also begun. The area is no longer grazed by livestock. 2006 photo by Charles E. Kay (5693-12). For more details see Plate 1332x on USU Extension's website.

Finally if repeat-photo projects are undertaken, all the photosets should be protected in clear, archival quality, plastic pages and all photos clearly identified. Arrangements also need to be made to properly archive the collection, so that researchers in the future can reoccupy the original camera stations; in other words, three-peat and four-peat photography (Klement and others 2001; Turner and others 2003). Our experience has been that these photographs should not be stored in agency files or at the bottom of some desk where they invariably will be lost as

people retire or are transferred. Instead, completed photosets should be deposited with institutions, whose mission it is to preserve historical material. We have made arrangements to permanently store our Fishlake repeat photos with Utah State University's Archives and Special Collections and our Dixie photos with the Archives and Special Collections at Southern Utah University. We have little doubt that future generations of ecologists and land managers will appreciate this foresight.

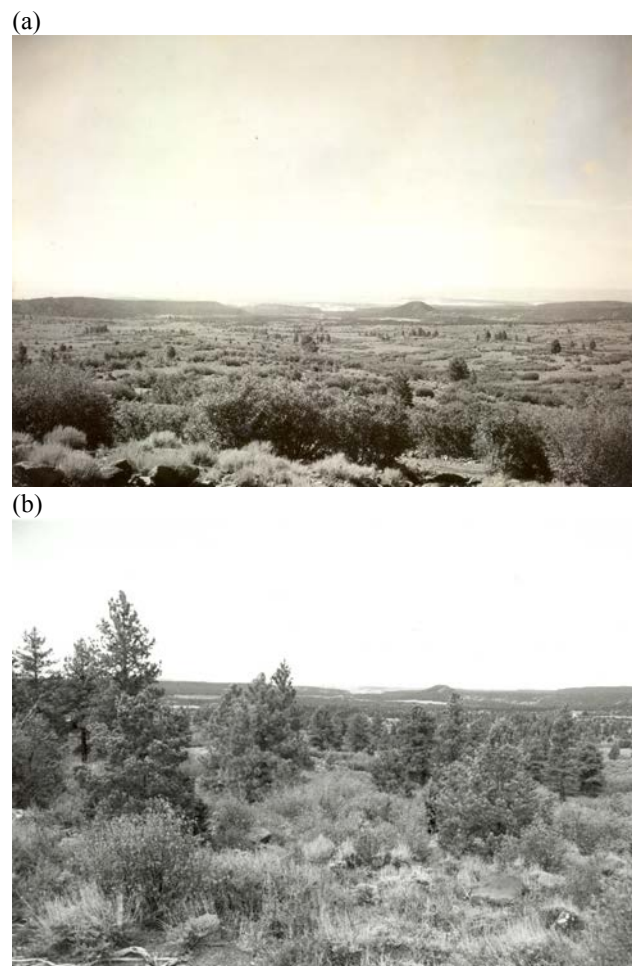


Figure 8—Vegetation change in the mountain brush zone above Boulder, Utah. The view is south to Ormand Point. The exact camera station could not be reoccupied due to the growth of woody vegetation but the retake was made as close as possible to the original camera station. The tall shrubs are mostly oakbrush (*Quercus gambelii*) and Utah serviceberry, while the lower-growing shrubs are primarily bitterbrush (*Purshia tridentata*), big sagebrush (*Artemisia tridentata*), and snowberry (*Symphoricarpos oreophilus*). (a) Ponderosa pine had begun to invade the mountain brush community when the 1936 image was made. Forest Service photo (307746). (b) That same site in 2002. Without fire the area is slowly changing to a pine forest. Photo by Charles E. Kay (4966-24). For more information see Plate 38x on USU Extension's website.

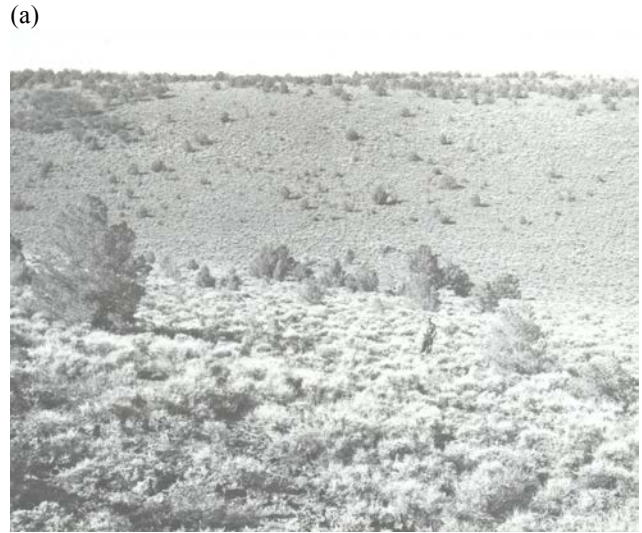


Figure 9—Vegetation change on the Parker Three-Step transect on Roundup Flat, Boulder Mountain, Utah. (a) The 1953 image shows a grass dominated site with few shrubs. At the time the allotment was grazed by domestic sheep. Forest Service photograph (unnumbered). (b) The allotment has since been converted to cattle and snowberry has invaded the site. 2003 photo by Charles E. Kay (5263-28). For additional details see Plate 362x on USU Extension's website.



Figure 10—The impact of pinyon-juniper invasion on mule deer winter range on the Dixie National Forest. Although it may be difficult to believe that the two photographs were taken from the exact same camera station, this is part of a Photo Plot Transect established by the Forest Service in 1943 and marked with steel stakes that still exist. As pinyon and juniper have increased, forage and shrub production have dramatically declined (Kay 2003). (a) 1943 image. Forest Service photograph (unnumbered). (b) The exact same camera station in 2005. Photo by Charles E. Kay (5516-15). For additional information see Plate 1033x on USU Extension's website.

ACKNOWLEDGEMENTS

This project was supported by mineral leasing funds provided by the Utah Legislature through Utah State University, as well as grants provided by the Utah Department of Agriculture and Food, the U.S. Forest Service, the Utah Division of Wildlife Resources Habitat Council, the Zion National History Association, and Utah State University Extension. We would also like to extend our appreciation to the many archivists and Forest Service officials, who helped locate photographs for this study, as well as the private property owners who allowed access to their lands so that we could reoccupy some original camera stations. Finally, we would like to thank Tom Hatch, Brad Johnson, Dennis Stowell, Leonard Blackham, Randy Simmons, Dave Thomas, Dave Grider, Marian Jacklin, Gretchen Merrill, Clint Reese, Evan Boshel, Kim Anderson, Kurt Robins, Bob Campbell, Dale Bartos, Kelly Fuhrmann, Dave Sharrow, Kristen Legg, Steve Robinson, Leslie Newkirk, and Kevin Heaton for their continued support and assistance on this project. The National Park Service provided research permits for Zion National Park, Bryce Canyon National Park, and Cedar Breaks National Monument. In 2004, we were able to hire Darrell McMahon to repeat photographs in the Pine Valley Mountains under a U.S. Department of Agriculture CSREES grant - - Project No. UTA-00556 - - to Dr. Randy Simmons. Borge Anderson Photography in Salt Lake City processed and printed the black and white film.

REFERENCES

- Anderson, M.K. 2005. *Tending the Wild: Native American knowledge and management of California's natural resources*. Berkely: University of California Press. 562p.
- Archer, S., Schimel, D.S., Holland, E.A. 1995. Mechanisms of shrubland expansion: land use, climate or CO₂? *Climatic Change* 29:91-99.
- Austin, D.D., Urness, P.J., Riggs, R.A. 1986. Vegetal change in the absence of livestock grazing, mountain brush zone, Utah. *Journal of Range Management* 39:514-517.
- Bond, W.J., Keeley, J.E. 2005. Fire as a global 'herbivore': the ecology and evolution of flammable ecosystems. *Trends in Ecology and Evolution* 20:387-394.
- Bond, W.J., Woodward, F.I., Midgley, G.F. 2005. The global distribution of ecosystems in a world without fire. *New Phytologist* 165:525-538.
- Bradford, D., Reed, F., LeValley, R.B. 2004. *When the grass stood stirrup-high: facts, photographs and myths of west-central Colorado*. Lawrence, KS: Allen Press. 182p.
- Branson, F.A. 1985. Vegetation changes on western rangelands. *Society for Range Management. Range Monograph* 2:1-76.
- Britz, M-L., Ward, D. 2007. Dynamic of woody vegetation in a semi-arid savanna, with a focus on brush encroachment. *African Journal of Range and Forage Science* 24:131-140.
- Burkhardt, J.W., Tisdale, E.W. 1976. Causes of juniper invasion in southwestern Idaho. *Ecology* 57:472-484.
- Christensen, E.M., Johnson, H.B. 1964. Presettlement vegetation and vegetation change in three valleys in central Utah. *Brigham Young University Science Bulletin Biological Series* 4(4):1-16.
- Cook, J.W., Brady, W.W., Aldon, E.F. 1992. Handbook for converting Parker loop frequency data to basal area. *Gen. Tech. Rep. RM-212*. Fort Collins, CO. Forest Service, Rocky Mountain Research Station. 22p.
- Cottam, W.P. 1947. Is Utah Sahara bound? *Bulletin of the University of Utah* 37(11):1-40.
- Cottam, W.P., Stewart, G. 1940. Plant succession as a result of grazing and meadow desiccation by erosion since settlement in 1862. *Journal of Forestry* 38:613-626.
- Crawford, J.A., Olson, R.A., West, N.E., Mosley, J.C., Schroeder, M.A., Whitson, T.D., Miller, R.F., Gregg, M.A., Boyd, C.S. 2004. Ecology and management of sage-grouse and sage-grouse habitat. *Journal of Range Management* 57:2-19.
- Creque, J.A., West, N.E., Dobrowolski, J.P. 1999. Methods in historical ecology: a case study of Tintic Valley, Utah. *In: Monson, Stephen B., Stevens, R. comps. 1999. Proceedings: ecology and management of pinyon-juniper communities within the interior West. Proceedings RMRS -P-9*. Fort Collins, CO. Rocky Mountain Research Station:121-133.
- Ffolliott, P.F., Gottfried, G.J. 2002. Dynamics of a pinyon-juniper stand in northern Arizona: a half-century history. *Res. Pap. RMRS-RP-35*. Fort Collins, CO. Forest Service, Rocky Mountain Research Station. 10p.
- Fowler, D.D. 1989. *Myself in the water: the western photographs of John K. Hillers*. Washington D.C.: Smithsonian Institution Press. 166p.
- Hall, F.C. 2002a. Photo point monitoring handbook: Part A - field procedures. *Gen. Tech. Rep. PNW-526*. Portland, OR. USDA, Forest Service, Pacific Northwest Research Station 48p.
- Hall, F.C. 2002b. Photo point monitoring handbook: Part B - concepts and analysis. *Gen. Tech. Rep. PNW-526*. Portland OR. USDA, Forest Service, Pacific Northwest Research Station. 86p.
- Hall, H.H., Cottam, W.P. 1955. The impact of man on the vegetation and soil of the Upper Valley area, Garfield County, Utah. *Proceedings of the Utah Academy of Science, Arts, and Letters* 32:105-106.
- Johnson, D.D., Miller, R.F. 2008. Intermountain presettlement juniper: distribution, abundance, and influence on postsettlement expansion. *Rangeland Ecology and Management* 61:82-92.
- Johnson, H.B., Polley, H.W., Mayeux, H.S. 1993. Increasing CO₂ and plant-plant interactions: effects on natural vegetation. *Vegetatio* 104/5:157-170.
- Johnson, W.M. 1969. Life expectancy of a sagebrush control in central Wyoming. *Journal of Range management* 22:177-182.
- Kay, C.E. 1995. Aboriginal overkill and native burning: implications for modern ecosystem management. *Western Journal of Applied Forestry* 10:121-126.
- Kay, C.E. 2003. Long-term vegetation change on Utah's Fishlake National Forest: a study in repeat-photography. *U.S. Forest Service and Utah State University Extension. 2003-53-084/42159 Region No. 8*. 175 p.
- Kay, C.E. 2006. Mule deer habitat: past and present. *Mule Deer Foundation Magazine No. 8 (Sept.-Oct.):22-26*.
- Kay, C.E. 2007. Are lightning fires unnatural? a comparison of aboriginal and lightning ignition rates in the United States. *Tall Timbers Fire Ecology Conference* 23:16-28.
- Klement, K.D., Heitschmidt, R.K., Kay, C.E. 2001. Eighty years of vegetation and landscape changes in the northern Great plains. *USDA, Agricultural Research Service Conservation Research Report* 45. 91p.
- Knapp, P.A., Soule, P.T. 1996. Vegetation change and the role of atmospheric CO₂ enrichment on a relict site in central Oregon:1960-1994. *Annals of the Association of American Geographers* 86:387-411.
- Knapp, P.A., Soule, P.T. 1998. Recent *Juniperus occidentalis* (western juniper) expansion on a protected site in central Oregon. *Global Change Biology* 4:347-357.
- LeFevre, L.H. 1973. *The Boulder County and its people: a history of the people of Boulder and the surrounding country - - one hundred years 1872-1973*. Springville: Art City Publishing. 294p.
- Lesica, P., Cooper, S.V., Kudray, G. 2007. Recovery of big sagebrush following fire in southwest Montana. *Rangeland Ecology and Management* 60:261-269.
- Magill, A.W. 1989. Monitoring environmental change with color slides. *Gen. Tech. Rep. PSW-117*. Berkely, CA. USDA, Forest Service, Pacific Southwest Forest Range Experiment Station. 55p.
- Miller, R.F., Rose, J.A. 1995. Historic expansion of *Juniperus occidentalis* in southeastern Idaho. *Great Basin Naturalist* 55:37-45.
- Miller, R.F., Tausch, R.J. 2001. The role of fire in juniper and pinyon woodlands: a descriptive analysis. *Tall Timbers Research Station Miscellaneous Publication* 11: 15-30.
- Miller, R.F., Tausch, R.J., McArthur, E.D., Johnson, D.D., Sanderson, S.C. 2008. Age structure and expansion of pinyon-juniper woodlands: a regional perspective in the Intermountain West. *Res. Pap. RMRS-RP-69*. Fort Collins, CO. Rocky Mountain Research Station. 15p.

- Miller, R.F., Wigand, P.E. 1994. Holocene changes in semiarid pinyon-juniper woodlands. *Bioscience* 44:465-474.
- Pyne, S.J. 1991. *Burning bush: a fire history of Australia*. New York: Henry Holt and Company. 520p.
- Reppert, J.N., Francis, R.E. 1973. Interpretation of trend in range condition from 3-step data. Res. Pap. RM-103. Fort Collins, CO. USDA, Forest Service, Rocky Mountain Research Station. 15p.
- Rogers, G.F. 1982. *Then and now: a photographic history of vegetation change in the central Great Basin desert*. Salt Lake City: University of Utah Press. 152 p.
- Rogers, G.F., Malde, H.E., Turner, R.M. 1984. *Bibliography of repeat photography for evaluating landscape change*. Salt Lake City: University of Utah Press. 179p.
- Roques, K.G., O'Connor, T.G., Watkinson, A.R. 2001. Dynamics of shrub encroachment in an African savanna: relative influences of fire, herbivory, rainfall and density dependence. *Journal of Applied Ecology* 38:268-280.
- Soule, P.T., Knapp, P.A. 1999. Western juniper expansion on adjacent disturbed and near-relict sites. *Journal of Range Management* 52:525-533.
- Soule, P.T., Knapp, P.A. 2000. *Juniper occidentalis* (western juniper) establishment history on two minimally disturbed research natural areas in central Oregon. *Western North America Naturalist* 60:26-33.
- Stewart, G. 1941. Historic records bearing on agricultural and grazing ecology in Utah. *Journal of Forestry* 39:362-375.
- Thilenius, J.F., Brown, G.R. 1974. Long-term effects of chemical control of big sagebrush. *Journal of Range Management* 27:223-224.
- Turner, R.M., Webb, R.H., Bowers, J.E., Hastings, J.R. 2003. The changing mile revisited. Tucson: University of Arizona Press. 334p.
- U.S. Forest Service. 1993. *Vegetation changes on the Manti-LaSal National Forest: a photographic study using comparative photographs from 1902-1992*. USDA, Forest Service, Price, UT. 128p.
- Vale, T.R., ed. 2002. *Fire, native peoples, and the natural landscape*. Washington D.C.: Island Press. 315p.
- Van Auken, O.W. 2000. Shrub invasions of North America semiarid grasslands. *Annual Review of Ecological Systems* 31:197-215.
- Wambolt, C.L., Payne, G.F. 1986. An 18-year comparison of control methods for Wyoming big sagebrush in southwestern Montana. *Journal of Range Management* 39:314-319.
- Watts, M.J., Wambolt, C.L. 1996. Long-term recovery of Wyoming big sagebrush after four treatments. *Journal of Environmental Management* 46:95-102.
- Woolsey, N.G. 1964. *The Escalante story: a history of the town of Escalante, and description of the surrounding territory*, Garfield County, Utah. Springville: Art City Publishing. 463p.
- Yorks, T.P., Wiest, N.E., Capels, K.M. 1994. Changes in pinyon-juniper woodlands in western Utah's Pine Valley between 1933-1989. *Journal of Range Management* 47:359-364.
- Young, J.A. 1989. Intermountain shrub-steppe plant communities: pristine and grazed. *National Wildlife Federation Scientific and Technical Series*. 12:3-14.

Classification and Management of Upland, Riparian, and Wetland Sites of USDI Bureau of Land Management's Miles City Field Office, Eastern Montana USA

Paul L. Hansen¹, William H. Thompson¹, Ray Smith², and Todd Yeager²

ABSTRACT

A system is presented for the classification and management of uplands, riparian, and wetland sites within the USDI Bureau of Land Management's Miles City Field Office in eastern Montana. The Miles City Field of the USDI Bureau of Land Management lies within the northern Great Plains and occupies approximately the eastern 1/3 of Montana. The concepts and terminology used in this document are consistent with usage proposed by Daubenmire (1952, 1968, 1978), and are used in numerous other vegetation-based ecological site classifications for North America. A dichotomous key utilizing indicator plant species is provided for field identification of the habitat types and major seral plant communities (for example community types) that are stable for time frames relevant to land management decisions. The habitat types (for example ecological site types) are identified first in the key. If this is not possible, the key then identifies the major seral community types. Within the description of each community type is a discussion of possible habitat types for the site. Each "type" includes detailed information for managing a particular site. The work utilized 1,126 sample plots resulting in a total of 96 different plant communities identified to either the habitat type (and phase) or community type level. There were 62 identified riparian/wetland types and 28 upland types. In addition, six types are types that can occur in both riparian/wetland and upland situations. There were a total of 663 unique species recorded for the 1,126 sampled stands. Of the 663 species, 551 (83 percent) were native, and 112 (17 percent) were introduced or contain an introduced element. The document can be downloaded from www.ecologicalsolutionsgroup.com.

INTRODUCTION

The vegetation of eastern Montana occupies an area of complex physiographic features, and diverse plant communities including upland, riparian, and wetland types. Land managers and scientists alike have long recognized the need to classify plant communities and landscapes, and have developed numerous forest and range (in other words, upland), riparian, and wetland classifications over the years. A number of vegetation-based ecological site classification and management documents have been developed for the region, including Cooper and Pfister (1985), Girard

and others (1989), Hansen and Hoffman (1988), Hansen and others (1984), Hansen and others (1995), Hoffman and Alexander (1987), Mueggler and Stewart (1980), Pfister and others (1977), Thompson and Hansen (2001), and Thompson and Hansen (2002).

Other, more limited works, included Mackie (1970) for a limited number of drainages in the Missouri River Breaks of north central Montana, Jorgensen (1979) for the Yellow Water Triangle of north central Montana, and Roberts (1980) of the Bear's Paw and Little Rocky Mountains of north central Montana.

This resulted in a number of documents covering different portions of the landscape in eastern Montana. The only document covering the entire study area was by Hansen and others (1995). However, this work only dealt with the riparian and wetland areas, and not the uplands. Large areas of the uplands were either not covered or incompletely covered by the numerous upland works. Therefore, the work by Hansen and others (2008) represents one of the first taxonomic (in other words, not just a list of types) vegetation-based ecological site classifications to cover an *entire* landscape of a large region, including the upland, riparian, and wetland sites.

The present study was started in the early fall of 2007 with the following objectives:

1. Develop one single vegetation-based ecological site classification and management document for all lands (in other words, uplands, riparian sites, and wetlands) within the administrative boundary of the USDI Bureau of Land Management's Miles City Field Office. The work would build from the earlier regional works of Cooper and Pfister (1985), Hansen and Hoffman (1988), Hansen and others (1984), Hansen and others (1995), Hoffman and Alexander (1987), Pfister and others (1977), Thompson and Hansen (2001), and Thompson and Hansen (2002), in addition to collecting new data to supplement these earlier works.
2. Relate the various types to the soils and climate of the area.
3. Present management information (where applicable) for the various habitat types and community types, including forage productivity, timber productivity, wildlife information, response to fire, rehabilitation

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Paul L. Hansen and William H. Thompson, Ecological Solutions Group LLC, Stevensville, MT

²Ray Smith and Todd Yeager, USDI Bureau of Land Management, Miles City Field Office, Miles City, MT

/restoration considerations, and recreational uses and considerations.

4. Determine successional relationships for the habitat types and major seral community types.

5. Relate the work to similar studies in this and adjacent regions.

PHYSIOGRAPHY AND GEOLOGY OF STUDY AREA

The study area is located in the eastern portion of Montana within the western part of the Great Plains Province. The Great Plains Province extends from the Dakotas into the eastern portions of Montana, Wyoming, and Colorado. The study area covers over 110,075 km² (42,500 mi²), including all or part of 17 of Montana's 56 counties. The USDI Bureau of Land Management's Miles City Field Office manages more than 1.1 million surface ha (2.7 million ac) of public land and over 5.0 million ha (12.4 million ac) of subsurface mineral estate in eastern Montana.

In the Miles City Field Office, the Great Plains Province is represented at the surface by broad basinal features. The basins within the Great Plains Province have accumulated sediments, such as sands, shales, and limestone, several miles in thickness. These basins form the source and reservoirs of Montana's fossil energy reserves of coal, crude oil, and natural gas. Large parts of two of these basins (the Williston Basin and the Powder River Basin) lying within this study area hold vast reserves of these fossil energy reserves, and continue to be the focus of energy development activities.

Two major river systems also dissect the eastern Montana landscape, the Missouri River and the Yellowstone River. The Missouri River roughly divides the northern third of the study area, flowing west to east from the Fort Peck Dam to Fort Union near the Montana-North Dakota border and the confluence of the Missouri and Yellowstone Rivers. The Missouri River receives significant additions to its flow from the Milk River, the Poplar River, and Big Muddy Creek, all of which flow from north to south into the parent stream. Average flows of the Missouri River equal approximately 312 m³/s (11,000 ft³/s) at Culbertson, Montana, approximately 32.2 km (20 mi) upstream of the Montana-North Dakota border.

The Yellowstone River flows northeast through the study area from Bighorn, Montana, past Sidney, Montana, near the Montana, South Dakota border. Major tributaries include the Tongue River and Powder River, both of which flow south to north into the parent stream. The average annual flow of the Yellowstone River, as gauged at Sidney, Montana, is approximately 368 m³/s (13,000 ft³/s).

The geology, soils, and climate of the region have produced a diverse mosaic of vegetation types on the landscape. The following photos attempt to show some of the vast diversity associated with this region (Figures 1 to 2).



Figure 1—Terry Badlands north of Terry, Montana.



Figure 2—Big Sheep Mountains in the fall north of Terry, Montana.

CLIMATE

The climate of eastern Montana is characterized as continental, having a long, cold winter and short, warm summers. For example, the highest temperature ever recorded in Montana was 47° C (117° F) at Medicine Lake on July 5, 1937, and at Glendive on July 20, 1893, both of which are in the study area. The coldest temperature ever recorded in Montana occurred in the Rocky Mountains (-57° C [-70° F]) at Rogers Pass. Miles City, which lies in the center of the study area, has seen temperatures as low as -39° C (-38° F) and as high as 45° C (113° F). The average temperature at Miles City is 9° C (49° F), with average highs in July of 31° C (88° F) and average lows in January of -8° C (17° F) (The Weather Channel 2008).

Average day of last frost in Miles City is May 14 and average day of first freeze is September 20, yielding a growing season of approximately 128 days. Average wind speed at Miles City is 15.6 km/h (9.7 mi/h), typically from the northwest. Miles City has an average precipitation of 34 cm (13.5 inch), with the most average precipitation occurring in June (6.2 cm [2.4 inch]) and the least in December (11.4 cm [0.5 inch]) (The Weather Channel 2008).

METHODS

Development of the Ecological Site Classification

We use the *habitat type* approach as outlined by Daubenmire (1952, 1968, 1970, 1978) in classifying riparian and wetland sites. We have adopted the *habitat type* approach for three main reasons:

1. While the habitat type is based on a climax plant association said to represent long-term biotic potential on the site, it also includes a series of seral stage communities, any one of which may occupy the site at some time in its history—even repeatedly following disturbance. Therefore, a site can be quite thoroughly described by naming its climax vegetation, as well as its current vegetation community. For example, a site may be described as being a *Fraxinus pennsylvanica/Prunus virginiana* (green ash/chokecherry) habitat type presently occupied by the *Populus deltoides/Symphoricarpos occidentalis* (Great Plains cottonwood/western snowberry) community type consisting of a mixed stand of old growth *Populus deltoides* (Great Plains cottonwood) with young *Fraxinus pennsylvanica* (green ash) saplings establishing in the shrub-dominated understory. The reader then is given a clear idea of the kind of site it is, the potential vegetation and the current vegetation.

2. The habitat type approach is primarily a vegetation-based, ecological site classification. A given habitat type may include a variety of soils types. A description of soils is included in a description of the habitat type, and soils information may be used to help characterize lower levels in the classification such as phases. Therefore, the habitat type approach uses vegetation as an integrator of the landscape and climatic conditions. This is especially appropriate in riparian zones where soils are commonly young and the pedogenic process is susceptible to frequent disruption by fluvial processes. Daubenmire noted that soil is a critical important ecological factor. However, he felt that vegetation responds to differences in moisture, fertility, temperature, and aeration rather than to parameters such as color, texture, structure, depth, sequence of horizons, and other soil features that are easily observed by the human eye.

3. The dynamics of edaphic and hydrologic conditions on the landscape typically cause the formation of complex mosaics of vegetation communities as intermixes of all stages of stand maturity and seral stage. We believe that the habitat type system of classifying sites offers the best presently available means for developing a working terminology for these natural places.

USE OF THE CLASSIFICATION

Overview

No two vegetation communities are exactly alike due to differences in climate, parent material, topography, elevation, disturbance regimes and a host of other variables. The ecological site classification concept used in this study is based on the premise that vegetation on a site reflects the culmination of all the elements at work there. Those species present on site best reflect the principal forces acting there.

This approach results in a classification that establishes baseline information for upland, riparian, and wetland sites within the USDI Bureau of Land Management Miles City Field Office administrative boundary. It provides land managers with information enabling them to develop appropriate management strategies and policies. Two features of this classification that increase its utility for land managers are the dichotomous key and the successional hierarchy established in the habitat type framework. The dichotomous key offers a simple approach to identifying vegetation types. Users, with some knowledge of the local flora, need only to identify the dominant overstory and understory species, and some indicator species, to use the classification. Type descriptions provide species lists, summarize physical parameters, outline successional trends, and present viable management strategies.

Another useful feature of the classification is the successional framework inherent within the habitat type concept. A habitat type represents the land area that supports, or has the potential to support, the same climax vegetation wherever it occurs (Daubenmire 1968). The climax vegetation, or plant association, represents the endpoint of succession on a site. Community types (intermediate, seral plant communities with similar floristic components in all structural layers) represent intermediate stages of succession. They can be organized within the framework of a habitat type system to form successional paths useful in predicting the pattern of community replacement from the pioneer to climax stages. Projection of potential, future vegetation communities in an area provides land managers with a tool for developing strategies and realistic goals.

Management Interpretations

Habitat type classifications provide a relatively permanent and ecologically-based system of land stratification in terms of vegetational potential (Daubenmire 1976). The habitat type is the basic unit in classifying land units or sites based on biotic potential, it emphasizes similarities in ecosystems which carry implications for a variety of land management objectives. Some of the practical implications of habitat type classification are in predicting livestock and wildlife forage production and wildlife habitat values, inventory, land type mapping, timber production, species selection for regeneration and/or rehabilitation, development of best management practices, growth rates of trees and shrubs, susceptibility of trees and shrubs to insects and disease, depth of soil moisture penetration, potential for producing browse after fire, soil management criteria, impacts of recreational uses, natural areas for preservation, downed woody fuels on the forest floor, and successional trends following disturbance. In addition, habitat types offer a basis of comparison and evaluation useful in designing and carrying out field experiments in ecology or applied natural resource disciplines.

There are typically three misconceptions about the use of habitat type classification: 1) the expectation of an abundance of climax vegetation to be present on the current landscape, 2) that natural resource managers need to manage solely for climax vegetation, and 3) that use of a habitat type classification requires climax or near-climax vegetation. For the first two misconceptions, the opposite is true. A very high percentage of our landscape has experienced disturbance resulting in domination by various seral stages. In the second case, preferred management strategies quite often favor seral, instead of climax potential species, regardless of habitat type. In the third misconception, comparing the relative reproductive success of certain plant species present, having known successional process, generally permits identification of the habitat type. In general, succession is more rapid for undergrowth species, providing insight into the habitat type for the site. Where stands have been severely disturbed, are in an early seral stage, or have a closed canopy resulting in a depauperate understory, comparison of the stand with nearby stands on physically similar sites can assist in habitat type identification.

Habitat type classification and management systems provide a critical part of the information needed to: describe the variety of vegetation communities potentially occupying an area; characterize the effect of disturbances or management on plant community distributions, threatened and endangered species, and other entities of conservation concern; identify realistic objectives and related management opportunities; provide a framework to

document successional relationships and seral communities; streamline monitoring design and facilitate extrapolation of monitoring interpretations; assess multi-resource potentials, capability, and suitability of management alternatives; provide a framework to help evaluate upland (in other words, forests and rangelands), riparian sites, and wetland health; assess risks for invasive species, fire, insects, disease, and flooding; conduct project planning and watershed analysis, and predict activity outcomes for a project or for land and resource management planning; allows planning of disturbed site vegetation rehabilitation/restoration to be based on site potential; and more effectively communicate with all stakeholders.

Developing Reasonable and Attainable Management Goals and Objectives

After upland, riparian, and wetland habitat types and community types have been identified on a site, there are several uses for the information. Understanding of the information available in this document will increase over time with use. Some examples of the uses of the information are discussed below.

Land management plans sometimes call for attaining certain vegetation communities. Using this document, an understanding of what vegetation is feasible for a site can be gained. For example, on a degraded site with only a scattering of *Artemisia tridentata* subsp. *wyomingensis* (Wyoming big sagebrush), the potential canopy cover can be learned from the canopy cover tables in the text. Specific objectives can then be written to increase the canopy cover on the site of *Artemisia tridentata* subsp. *wyomingensis* (Wyoming big sagebrush) from 3 percent to 20 percent. The canopy cover tables can be used to write species lists for site rehabilitation or restoration projects. For example, on oil or gas pad, there may be limited vegetation remaining. The canopy cover tables provide a list of species that could be on the site. The species that are most desirable, available for planting, and easiest to obtain or establish can be selected.

If the goal is to provide shading of a stream for fish habitat, the tables will provide a list of species that will grow on a type, so that informed decisions can be made to avoid planting species unsuited for the purpose. Long-term planning, land use decisions, threatened and endangered species consultations, and environmental documents can be guided by successional information present in each habitat type or community type described.

Therefore, habitat type classification systems are useful to land and resource managers by providing: a permanent and ecologically based system of site classification that is referenced to vegetation potential (Daubenmire 1976); a

vegetational classification for near-climax to climax communities; a way in which successional stages can be identified and described, thereby increasing our knowledge and ability to predict change on the landscape; and a basis for predicting results of management decisions or expected trends resulting from natural disturbances.

OVERALL RESULTS

The work resulted in a total of 96 different plant communities identified to either the habitat type (and phase) or community type level. A taxonomic key is included to key to the various types, along with an appendix showing photos/illustrations of the 86 indicator species used in the key (trees = 14; shrubs = 24; graminoids = 41; and forbs = 7). Each indicator species is discussed in terms of its family, origin status (native vs. introduced), habit, habitat/distribution, inflorescence, stems, leaves, and similar species.

In the text, each habitat type or community type contains a detailed discussion on the ecology and management of the type including: number of stands sampled; distribution within the study area; the status of the type (in other words, riparian/wetland type or upland type); location on the landscape; vegetation of late seral to climax stands, and disturbed and/or early to mid seral stands (including canopy cover, range of canopy cover, constancy, prominence index,

and origin status (native vs. introduced); successional information; soils; livestock management information (forage production, species palatability, etc.); timber management information (if appropriate) (for coniferous and some deciduous types—tree population analysis of stand data, timber productivity [basal area, site index, cumulative mean annual increment, etc.]); wildlife management information; fisheries management information (if appropriate); fire management information; rehabilitation/restoration considerations; recreational uses and other considerations; and other studies.

Table 1 shows the break down by type (in other words, coniferous forest, deciduous forest, shrub, graminoid, and forb types) of the 1,126 stands sampled.

Table 2 shows the break down of the 96 different plant communities by riparian/wetland (62) or upland types (28). Six types are types that can occur in both riparian/wetland and upland situations.

Table 3 gives the overall break down by lifeform (tree, shrub, graminoid, forb, and ferns and allies) of species recorded throughout the entire study. There were a total of 663 unique species (in other words, species richness) recorded for the 1,126 sampled stands. Of the 663 species, 551 (83 percent) are native, and 112 (17 percent) were introduced or contain an introduced element.

Table 1—Number of stands and number of habitat types (h.t.)/community types (c.t.) by lifeform.

Type	Number of Stands	Number of h.t.s/phases	Number of c.t.'s
Coniferous Forest Types (h.t./c.t.)	108	13	0
Deciduous Forest Types (h.t./c.t.)	265	5	15
Shrub Types (h.t./c.t.)	337	13	14
Graminoid Types (h.t./c.t.)	347	24	8
Forb Types (h.t./c.t.)	69	1	3
Totals	1,126	56	40

Table 2—Number of stands, number of riparian/wetland types, and number of upland types by lifeform.

Type	Number of Stands	Number of Riparian / Wetland Types	Number of Upland Types	Both ^a
Coniferous Forest Types (h.t./c.t.)	108	2	9	2
Deciduous Forest Types (h.t./c.t.)	265	18	1	1
Shrub Types (h.t./c.t.)	337	15	10	2
Graminoid Types (h.t./c.t.)	347	23	8	1
Forb Types (h.t./c.t.)	69	4	0	0
Totals	1,126	62	28	6

^aBoth—Refers to habitat types or community types that can occur in both riparian/wetland and upland situations. The probability of occurrence in either a riparian/wetland site or an upland site is discussed in the write-up on each habitat type or community type.

Table 3—Total number of unique species by lifeform and by native, introduced, or both for the 1,126 sampled stands.

Lifeform	Number	Native ^a	Introduced ^b	Both ^c
Trees	14	13	1	0
Shrubs	77	72	2	3
Graminoids	140	116	17	7
Forbs	422	342	60	20
Ferns and Allies	10	8	0	2
Totals	663	551	80	32

^aNative = native to pre-Columbian North America.

^bIntroduced = introduced by post-Columbian human immigrants.

^cBoth = species contains native and introduced elements (**NOTE:** Those plant specimens only identified to genus and the genus includes both native and introduced species, were identified as “Both.”)

Hansen and others (2008) presents a table (table 6; pages 35 to 41) showing *species richness (in other words, the total number of plant species recorded in stands sampled)* for each habitat type and community type. The table also shows additional information on total number of species per lifeform. The table includes only those types (h.t./c.t.) that had a large enough sample size to include a summary table in the description of the type. Some general trends are as follows:

- Disturbance generally causes an increase in species richness.
- The smaller the sample size, the lower the species richness. Therefore, caution should be exercised when making comparison to stands with a limited sample size.
- Woody-dominated types tend to have a greater overall species richness than herbaceous-dominated types. There are exceptions to this trend, such as the *Andropogon scoparius/Carex filifolia* (little bluestem/threadleaf sedge) habitat type with a species richness of 126 and a total sample size of 29 stands.
- Types found on mesic sites tend to have the greatest species richness, with those found on either dry sites or the wet sites having lower species richness. Many of the wetland herbaceous types are monospecific in undisturbed sites, and upon disturbance, the species richness increases, generally due to invasion by weedy species. An example is the *Scirpus pungens* (three-square bulrush) habitat type that has a species richness of 5 in relatively undisturbed stands ($N = 6$), and a species richness of 44 in disturbed stands ($N = 5$).

Some specific results are as follows:

Coniferous Forest Types

Highest species richness = 106 species (*Pinus ponderosa/Prunus virginiana* [ponderosa pine / chokecherry] habitat type, the *Berberis repens* [Oregon grape] phase; riparian/wetland type; number of stands sampled = 14).

Lowest species richness = 22 species (*Pseudotsuga menziesii/Juniperus scopulorum* [Douglas fir/Rocky Mountain juniper] habitat type; upland type; number of stands sampled = 2).

Deciduous Forest Types

Highest species richness = 197 species (disturbed and/or early to mid-seral stands of the *Fraxinus pennsylvanica/Prunus virginiana* [green ash/chokecherry] habitat type; riparian/wetland type; number of stands sampled = 90).

Lowest species richness = 27 species (*Elaeagnus angustifolia* [Russian olive] community type; riparian/wetland type; number of stands sampled = 3).

Shrub Types

Highest species richness = 116 species (disturbed and/or early to mid-seral stands of the *Artemisia cana* subsp. *cana/Agropyron smithii* [plains silver sagebrush/western wheatgrass] habitat type; riparian/wetland type; number of stands sampled = 31).

Lowest species richness = 18 species (*Tamarix chinensis* [salt cedar] community type; riparian/wetland type; number of stands sampled = 3).

Graminoid Types

Highest species richness = 126 species (*Andropogon scoparius/Carex filifolia* [little bluestem/threadleaf sedge] habitat type; upland type; number of stands sampled = 29).

Lowest species richness = 5 species (late seral to climax stands of the *Scirpus pungens* [three-square bulrush] habitat type; riparian/wetland type; number of stands sampled = 6).

Forb Types

Highest species richness = 26 species (*Glycyrrhiza lepidota* [American licorice] community type; riparian/wetland type; number of stands sampled = 4).

Lowest species richness = 8 species (*Polygonum amphibium* [water smartweed] community type; riparian/wetland type; number of stands sampled = 7).

Hansen and others (2008) also presents a table (table 7; pages 42 to 48) showing the average number of species of trees, shrubs, graminoids, forbs, ferns/allies, and **average number of individual species per stand sampled** by habitat type (h.t.) and community type (c.t.). The table includes only those types (h.t./c.t.) that had a large enough sample size to include a summary table in the description of the type. Once again, some general trends are as follows:

- Disturbance generally causes an increase in the average number of species per stand.
- The smaller the sample size, the lower the average number of species per stand. Therefore, caution should be exercised when making comparison to stands with a limited sample size.
- Woody-dominated types tend to have a greater overall average number of species per stand than herbaceous-dominated types. Once again, there are exceptions to this trend, such as the *Agropyron spicatum*/*Bouteloua curtipendula* (bluebunch wheatgrass/sideoats grama) habitat type with an average number of species per stand of 32.1 and a total sample size of 4 stands.
- Types found on mesic sites tend to have the largest average number of species per stand, with those found on either dry sites or the wet sites having lower diversity. Many of the wetland herbaceous types are monospecific in undisturbed sites, and upon disturbance, the average number of species per stand increases, generally due to invasion by weedy species. An example is the *Scirpus pungens* (three-square bulrush) habitat type that has an average number of species per stand of 1.6 in relatively undisturbed stands ($N = 6$), and a species richness of 7.4 in disturbed stands ($N = 5$).

Some specific results are as follows:

Coniferous Forest Types

Highest average number of species per stand = 29.0 species (*Juniperus scopulorum*/*Agropyron spicatum* [Rocky Mountain juniper/bluebunch wheatgrass] habitat type; upland type; number of stands sampled = 5).

Lowest average number of species per stand = 14.0 species (*Juniperus scopulorum*/*Cornus stolonifera* [Rocky Mountain juniper/red-osier dogwood] habitat type and the *Pseudotsuga menziesii*/*Juniperus scopulorum* [Douglas fir/Rocky Mountain juniper] habitat type; riparian/wetland type and upland type respectively; number of stands sampled = 6 and 2 respectively).

Deciduous Forest Types

Highest average number of species per stand = 34.2 species (*Populus tremuloides*/*Berberis repens* [quaking aspen/Oregon grape] habitat type; riparian/wetland type; number of stands sampled = 4).

Lowest average number of species per stand = 7.2 species (*Populus deltoides*/Recent Alluvial Bar [Great Plains cottonwood/Recent Alluvial Bar] community type; riparian/wetland type; number of stands sampled = 13).

Shrub Types

Highest average number of species per stand = 34.2 species (*Juniperus horizontalis*/*Carex heliophila* [creeping juniper/sun sedge] habitat type; upland type; number of stands sampled = 6).

Lowest average number of species per stand = 6.1 species (undisturbed stands of the *Salix exigua* [sandbar willow] community type; riparian/wetland type; number of stands sampled = 17).

Graminoid Types

Highest average number of species per stand = 32.1 species (*Agropyron spicatum*/*Bouteloua curtipendula* [bluebunch wheatgrass/sideoats grama] habitat type; upland type; number of stands sampled = 4).

Lowest average number of species per stand = 1.5 species (late seral to climax stands of the *Scirpus acutus* [hardstem bulrush] habitat type; riparian/wetland type; number of stands sampled = 16).

Forb Types

Highest average number of species per stand = 8.3 species (*Glycyrrhiza lepidota* [American licorice] community type; riparian/wetland type; number of stands sampled = 4).

Lowest average number of species per stand = 1.4 species (late seral to climax stands of the *Typha latifolia* [common cattail] habitat type; riparian/wetland type; number of stands sampled = 36).

Additional information on average number of species per lifeform is also presented in the table.

Finally, Hansen and others (2008) presents a table (table 8; pages 49 to 55) showing the average number of species of trees, shrubs, graminoids, forbs, ferns/allies, and **average canopy cover of each lifeform group per stand sampled** by habitat type (h.t.) and community type (c.t.). The table includes only those types (h.t./c.t.) that had a large enough sample size to include a summary table in the description of

the type. This enables a quick comparison of structural complexity among the types described in the study area.

Some general trends are as follows:

- Disturbance generally causes an increase in the average canopy cover of a stand.
- Woody-dominated types tend to have a greater overall average canopy cover per stand than herbaceous-dominated types. Once again, there are exceptions to this trend, such as the stands of the late seral to climax *Stipa comata*/*Carex heliophila* (needle-and-thread/sun sedge) habitat type with an average canopy cover per stand of 200.0 percent and a total sample size of 4 stands.
- Types found on mesic sites tend to have the largest average canopy cover per stand, with those found on either dry sites or the wet sites having lower amounts. Many of the wetland herbaceous types are monospecific in undisturbed sites, and upon disturbance, the average number of species per stand increases resulting in an increase in average canopy cover, generally due to invasion by weedy species. An example is the *Carex rostrata* (beaked sedge) habitat type, *Carex rostrata* (beaked sedge) phase that has an average canopy cover per stand of 97.1 percent in relatively undisturbed stands ($N = 12$), and an average canopy cover per stand of 126.9 percent in disturbed stands ($N = 4$).

Some specific results are as follows:

Coniferous Forest Types

Highest average canopy cover per stand = 325.4 percent (*Pinus ponderosa*/*Cornus stolonifera* [ponderosa pine/red-osier dogwood] habitat type; riparian/wetland type; number of stands sampled = 4).

Lowest average canopy cover per stand = 93.2 percent (*Pinus flexilis*/*Agropyron spicatum* [limber pine/bluebunch wheatgrass] habitat type; upland type; number of stands sampled = 8).

Deciduous Forest Types

Highest average canopy cover per stand = 356.5 percent (*Elaeagnus angustifolia* [Russian olive] community type; riparian/wetland type; number of stands sampled = 3).

Lowest average canopy cover per stand = 68.7 percent [*Populus deltoides*/Recent Alluvial Bar [Great Plains cottonwood/Recent Alluvial Bar] community type; riparian/wetland type; number of stands sampled = 13).

Shrub Types

Highest average canopy cover per stand = 227.8 percent (*Shepherdia argentea* [silver buffaloberry] community type; riparian/wetland type; number of stands sampled = 27).

Lowest average canopy cover per stand = 39.3 percent (*Sarcobatus vermiculatus*/*Agropyron spicatum* [greasewood/bluebunch wheatgrass] habitat type; upland type; number of stands sampled = 3).

Graminoid Types

Highest average canopy cover per stand = 200.0 percent (*Stipa comata*/*Carex heliophila* [needle-and-thread/sun sedge] habitat type; upland type; number of stands sampled = 4).

Lowest average canopy cover per stand = 84.0 percent (late seral to climax stands of the *Spartina pectinata* [prairie cordgrass] habitat type; riparian/wetland type; number of stands sampled = 6).

Forb Types

Highest average canopy cover per stand = 134.1 percent (*Glycyrrhiza lepidota* [American licorice] community type; riparian/wetland type; number of stands sampled = 4).

Lowest average canopy cover per stand = 73.9 percent (*Salicornia rubra* [red glasswort] community type; riparian/wetland type; number of stands sampled = 9).

Additional information on average number of species and average canopy cover per lifeform is also presented in the table.

We also have a section on coniferous timber productivity, which includes distribution of the conifer types along a moisture gradient, tree population analysis, and timber productivity (site index, basal area, culmination of mean annual increment [a measure of tree growth]).

In addition, we have included discussions on number of plant species commonly found in the study area. These species typically present challenges to land managers. Some are native species, while others are invasive, weedy species. We have attempted to explain the ecology of these species to better interpret and understand their role on the landscape. Our hope is that this information will assist land managers in making informed decisions. The plant species are: *Juniperus scopulorum* (Rocky Mountain juniper)—ecological considerations, water use, habitat values; *Artemisia tridentata* subsp. *wyomingensis* (Wyoming big sagebrush)—ecological considerations, fire effects, palatability, wildlife uses, rehabilitation/restoration considerations; *Artemisia cana* subsp. *cana* (plains silver sagebrush)—ecological considerations, fire effects, palatability, wildlife uses, rehabilitation/restoration considerations; *Sarcobatus vermiculatus* (greasewood)—ecological considerations, fire effects, palatability, wildlife uses, rehabilitation/restoration considerations; annual

Bromus (brome) species such as *Bromus japonicus* (Japanese brome) and *Bromus tectorum* (cheatgrass)—species descriptions, physical site requirements, ecological considerations, forage values, fire adaptations; and *Elaeagnus angustifolia* (Russian olive), *Tamarix chinensis* (salt cedar), and *Agropyron cristatum* (crested wheatgrass).

REFERENCES

- Cooper, S.V., Pfister, R.D. 1985. Forest habitat types of the Crow and Northern Cheyenne Indian Reservations. Final Report. Ogden, UT: USDA, Forest Service, Intermountain Forest and Range Experiment Station. 103 p.
- Daubenmire, R.D. 1952. Forest vegetation of northern Idaho and adjacent Washington, and its bearing on concepts of vegetation classification. *Ecological Monographs* 22: 301-330.
- Daubenmire, R.D. 1968. Plant communities. New York: Harper and Row. 300 p.
- Daubenmire, R.D. 1970. Steppe vegetation of Washington. *Tech. Bull.* 62. Washington State University, Pullman, WA: Washington Agricultural Experiment Station. 131 p.
- Daubenmire, R.D. 1976. The use of vegetation in assessing the productivity of forest lands. *Botanical Review* 42(2): 115-143.
- Daubenmire, R.D. 1978. Plant geography with special reference to North America. New York: Academic Press. 338 p.
- Girard, M.M., Goetz, H., Bjugstad, A.J. 1989. Native woodland habitat types of southwestern North Dakota. *Res. Pap. RM-281*. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Forest and Range Experiment Station. 36 p.
- Hansen, P.L., Hoffman, G.R. 1988. The vegetation of the Grand River/Cedar River, Sioux, and Ashland Districts of the Custer National Forest: a habitat type classification. *Gen. Tech. Rep. RM-147*. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Forest and Range Experiment Station. 68 p.
- Hansen, P.L., Hoffman, G.R., Bjugstad, A.J. 1984. The vegetation of Theodore Roosevelt National Park, North Dakota: a habitat type classification. *Gen. Tech. Rep. RM-113*. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Forest and Range Experiment Station. 35 p.
- Hansen, P.L., Pfister, R.D., Boggs, K., Cook, B.J., Joy, J., Hinckley, D.K. 1995. Classification and management of Montana's riparian and wetland sites. *Misc. Pub. No. 54*. Missoula, MT: University of Montana, School of Forestry, Montana Forest and Conservation Experiment Station. 646 p.
- Hansen, P.L., Thompson, W.H., Massey, J.G., Thompson, M. 2008. Classification and management of upland, riparian, and wetland sites of USDI Bureau of Land Management's Miles City Field Office, eastern Montana USA. Final contract report to the USDI Bureau of Land Management Miles City Field Office. Stevensville, MT: Ecological Solutions Group LLC. 640 p. plus 91 p. of Appendix B (Indicator Species).
- Hoffman, G.R., Alexander, R.R. 1987. Forest vegetation of the Black Hills National Forest in western South Dakota and eastern Wyoming: a habitat type classification. *Res. Pap. RM-270*. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Forest and Range Experiment Station. 48 p.
- Jorgensen, H.E. 1979. Vegetation of the Yellow Water Triangle, Montana. Billings, MT: Montana Department of Fish and Game/USDI Bureau of Land Management Unnumbered Publication. 57 p.
- Mackie, R.J. 1970. Range ecology and relations of mule deer, elk, and cattle in the Missouri River Breaks, Montana. Bethesda, MD: The Wildlife Society, *Wildlife Monograph* 20. 79 p.
- Mueggler, W.F., Stewart, W.L. 1980. Grassland and shrubland habitat types of western Montana. *Gen. Tech. Rep. INT-66*. Ogden, UT: Forest Service, Intermountain Forest and Range Experiment Station. 154 p.
- Pfister, R.D., Kovalchik, B.L., Arno, S.F., Presby, R.C. 1977. Forest habitat types of Montana. *Gen. Tech. Rep. INT-34*. Ogden, UT: Forest Service, Intermountain Forest and Range Experiment Station. 175 p.
- Roberts, D.W. 1980. Forest habitat types of the Bear's Paw Mountains and Little Rocky Mountains, Montana. Missoula, MT: University of Montana. 115 p. Thesis.
- The Weather Channel. 2008. Available: <http://www.weather.com> [10 June 2008].
- Thompson, W.H., Hansen, P.L. 2001. Classification and management of riparian and wetland sites of the Saskatchewan prairie ecozone and parts of adjacent subregions. Final contract report to the Saskatchewan Wetland Conservation Corporation. Corvallis, MT: Bitterroot Restoration, Inc. 298 p.
- Thompson, W.H., Hansen, P.L. 2002. Classification and management of riparian and wetland sites of the Alberta Grassland Natural Region and adjacent subregions. Final contract report to the Alberta Riparian Habitat Management Program-Cows and Fish. Corvallis, MT: Bitterroot Restoration, Inc. 416 p.

Ecological Assessment of Sagebrush Grasslands in Eastern Wyoming

Amy C. Ganguli¹, Jonathan B. Haufler¹, Carolyn A. Mehl¹, and Scott D. Yeats¹

ABSTRACT

An understanding of existing ecosystem conditions is necessary for planning efforts that include formulation of landscape conservation goals and implementation strategies. In support of a landscape planning effort for a 946,000-ac mixed-ownership area in eastern Wyoming, we used remote sensing and field sampling to assess existing ecosystem conditions of terrestrial ecological sites. We used SPOT 5, 33-ft (10-m) multi-spectral satellite imagery combined with NRCS ecological sites to create a geographic information system layer of vegetation cover by ecological site. We then integrated the remote sensing information with field data (571 plots) collected from a stratified random design from 2003 through 2005. The integration of the field data with the satellite mapping provided specific information about each terrestrial ecological site including species composition, productivity, structure, and shrub cover. Western wheatgrass was the most dominant species across all of the terrestrial ecological sites followed by big sagebrush, needle and thread, blue grama, annual brome species and to a lesser extent threadleaf sedge, and six weeks fescue. We found species that typically decrease with grazing (for example green needlegrass, bluebunch wheatgrass, Indian ricegrass) to be lacking or entirely absent from plant communities. Introduced species, especially the annual bromes, were prevalent across all ecological sites. Over 55 percent of the terrestrial ecosystems we sampled had greater than five percent relative cover of introduced plant species. Current ecosystem conditions for many wildlife of the area, as identified by our assessment, had generally lower habitat quality than desired and treatments to improve these conditions are planned.

INTRODUCTION

A clear understanding of existing ecosystem conditions is necessary to formulate conservation objectives that include managing for ecosystem diversity and wildlife in working landscapes. The Thunder Basin region, located in eastern Wyoming (figure 1), is recognized as one of the most ecologically significant grasslands in the United States (Forrest and others 2004). The region supports plant and animal communities that include grassland and sagebrush obligate species that have been identified as species of concern. Land ownership is mixed, with a majority of the land in private ownership, but with substantial public lands including lands of the Thunder Basin National Grasslands, Bureau of Land Management, and state of Wyoming.

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Amy C. Ganguli, Jonathan B. Haufler, Carolyn A. Mehl, and Scott D. Yeats, The Ecosystem Management Research Institute, Seeley Lake, MT

Although the dominant land use in Thunder Basin is ranching, energy production plays a key economic role. In fact, some of the largest coal mines in the world are located in Thunder Basin and the area also supports substantial oil and gas production.

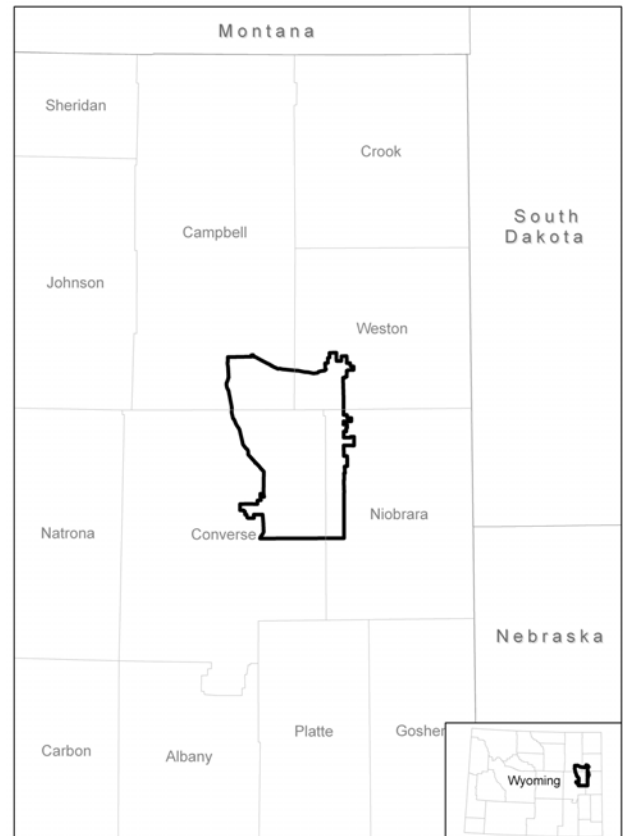


Figure 1—Location of the Thunder Basin planning area in Converse, Campbell, Niobrara, and Weston Counties of eastern Wyoming.

Thunder Basin is often identified as a key area of native grasslands, having substantial areas that have not been tilled or converted to other uses. However, to understand the actual conditions present in the landscape, an assessment of actual plant communities that currently exist was needed. The dominant land uses of the landscape are ranching and energy production, which have resulted in different disturbance pressures placed on the landscape than what occurred historically. The ecosystem diversity of this landscape prior to European settlement was primarily a function of the different types of soils, other site factors, as well as historical disturbance regimes. The primary

historical disturbances were fire, grazing especially by bison, and drought events (Knight 1994). Although wildfires still occur, and a few ranchers have tried an occasional prescribed burn, fire suppression activities and other landscape conditions have substantially reduced the role of fire in the landscape (Fisher and others 1987; Perryman 1996; Perryman and Laycock 2000). Grazing today is by cattle, sheep, and horses, with relatively uniform grazing levels applied across the landscape as compared with estimates of historical grazing patterns and pressure of bison (Ganguli and others 2008; Fuhlendorf and Engle 2001). Hydrology within the planning area has been altered by construction of numerous water catchments. In addition, some streams have been re-routed to provide irrigation and in some cases water has been added to streams as a byproduct of energy production activities. Infrastructure for transportation has been established including railroad and roads. Another modern feature of the Thunder Basin landscape is the presence of introduced plant species.

As part of a landscape assessment aimed at identifying and quantifying the cumulative changes to native terrestrial ecosystem diversity within the Thunder Basin planning area, the goal of this study was to identify existing terrestrial ecosystem conditions. We used remote sensing and conducted plant community field sampling in the 946,000 ac planning area from 2003 through 2005 to characterize the existing terrestrial ecosystem conditions present within the Thunder Basin planning area.

STUDY AREA

This study was located in a portion of the Thunder Basin National Grasslands in northeastern Wyoming (figure 1). The area is characterized by tablelands and high open hills with occasional rock outcrops and elevations ranging from 4,000 to 5,300 ft (1,219 to 1,615 m). Bedrock geology of the area is dominated by sandstone and shale from the Fort Union and Wasatch formations. Climate of the Thunder Basin region is broadly characterized as interior continental with hot summers and cold winters. Average annual precipitation is 12.4 in (31.5 cm) with approximately 80 percent of the precipitation each year falling between April and October as rain. Terrestrial ecological sites occurring in the study area include loamy (268,298 ac; 108,578 ha), shallow clayey (224,360 ac; 90,796 ha), clayey (125,637 ac; 50,844 ha), sands/sandy (64,683 ac; 26,177 ha), shallow loamy (62,541 ac; 25,310 ha), saline upland (45,902 ac; 18,576 ha), badlands/gullied land (35,560 ac; 14,391 ha), shallow sandy (35,190 ac; 14,241 ha), shallow hilly (30,269 ac; 12,250 ha), and very shallow (2,791 ac; 1,129 ha).

METHODS

We used a combination of remote sensing and field sampling to assess the existing ecosystem conditions of the TBGPEA planning area. Remote sensing using SPOT 5 multi-spectral satellite imagery was combined with NRCS delineated ecological sites to create a geographic information system layer of vegetation classes by ecological site. The remote sensing was used to broadly characterize the landscape into land cover classes of interest, such as plant communities with greater or less than 10 percent big sagebrush cover, riparian zones, ponderosa pine woodlands, sparsely vegetated areas, and disturbed areas. We then integrated this information with field data sampled from 2003 through 2005. Field sampling provided specific information about the existing vegetation conditions of each combination of vegetation and ecological site.

Remote Sensing Vegetation Classification

We used 33ft (10 m) resolution multi-spectral imagery from the SPOT 5 satellite to classify the TBGPEA planning landscape into vegetation classes. We obtained SPOT 5 satellite images of the Thunder Basin area from iCubed Technologies (Fort Collins, CO), who worked in conjunction with the SPOT Corporation for image acquisition. Image interpretation was conducted by Symmetry, Inc. The interpretation used image analysis software to perform both unsupervised and supervised image classifications. The unsupervised classification resulted in each 33ft (10 m) pixel being assigned to a class based on spectral similarity. The supervised classification then used the unsupervised classification and assigned these supervised classes to a land cover class that we identified based on training points delineated in the field. Following classification of the SPOT image into land cover types we assessed image classification accuracy by randomly selecting ground-truthing sample points in ArcGIS. Sample points for ground-truthing were selected by land cover class and involved field measurements of shrub cover and general observations of site conditions characteristics during the 2006 growing season. SPOT 5 classification accuracy was summarized in an omission/commission matrix. We then stratified the SPOT 5 classified imagery by ecological site to obtain a landscape characterization of dominant land cover classes.

Vegetation and Ecological Site Sampling

We conducted vegetation sampling in the Thunder Basin planning area from 2003 through 2005. Sampling plots were distributed on public lands and participating private lands using a stratified random design. We stratified across

NRCS ecological sites as well as dominance of shrub cover, dominance of ponderosa pine, and dominance of prairie dog towns. Mapping errors or inconsistencies identified in the field were subsequently corrected in our analysis of plant community existing conditions.

We sampled the vegetation on each plot for cover, productivity, and structure and used the PLANTS database as a taxonomic reference (USDA, NRCS 2006). Sampling crews were trained in the same way each year to provide for consistent application of methods. A 164-ft (50 m) vegetation transect was placed in a randomly determined direction from the designated plot location and 15 quadrats, 8 in x 20 in (20 cm x 50 cm) were sampled. Canopy cover of plant species occurring in the quadrats was estimated using the following cover classes (0.01 to 1 percent, 1 to 10 percent, 10 to 20 percent, 20 to 30 percent, 30 to 40 percent, 40 to 50 percent, 50 to 60 percent, 60 to 70 percent, 70 to 80 percent, 80 to 90 percent, 90 to 99 percent, and 99 to 100 percent). For subsequent analysis we used cover class midpoints and converted absolute cover values to relative cover values for each plot and used these values in our assessment of existing conditions. Productivity of grasses and forbs were estimated by species on the first 10 quadrats sampled using double sampling (Pechanec and others 1937). Estimates of plant wet weight were made only on plants rooted in the microplots and were converted to dry matter estimates using conversion tables based on plant phenology (USDA/NRCS 2003). In addition, we recorded the canopy cover of species not encountered in the quadrat sampling in a 82.0 ft x 49.2 ft (25 m x 15 m) or 4,036 ft² (375 m²) area. Canopy cover of all species of woody vegetation was sampled along the 164 ft (50 m) transect using the line intercept method (Canfield and others 1941). Vegetation structure was determined by making 12 Robel pole measurements along the plot transect. Robel measurements were made by determining the highest 1-in (2.5 cm) band on the Robel pole that was totally or partially visible when viewed from a distance of 13.1ft (4 m) and a height of 3.3ft (1 m) (Robel 1970).

RESULTS

Remote Sensing Vegetation Classification

Accuracy assessment of the SPOT 5 land cover classification indicated that the overall accuracy was 81.6 percent (table 1). The riparian, water, sparsely vegetated, and disturbed cover classes had the highest classification accuracy. Due to the timing of SPOT 5 image acquisition, greasewood presence could not be distinguished. That is, the spectral reflectance of greasewood could not be distinguished from the surrounding vegetation. The

percentage of sagebrush and other shrub cover occurred along a continuous gradient, making the determination of <10 percent and >10 percent categories difficult. Correct classification of grassland (in other words, shrub cover <10 percent) was 54 percent and grassland with shrubs (in other words, shrub cover >10 percent) was 73 percent, with most classification errors resulting from misclassification of the amount of sagebrush. Generally, accuracy of the detection of stands with either high amounts of sagebrush or low amounts of sagebrush was generally good, but the intermediate levels of sagebrush were more difficult to distinguish. Ponderosa pine and other cover classes were mapped with relatively high accuracy.

The numbers of ac found in each land cover class are summarized by Ecological Site and are presented in table 2. Grasslands and shrublands occupied 68.8 percent of the terrestrial ecosystems in the Thunder Basin planning area. Further examination of the big sagebrush cover for grasslands and shrublands on the primary terrestrial Ecological Sites (in other words, Clayey, Shallow Clayey, Loamy, Shallow Loamy, Sands/Sandy, and Shallow Sandy), which occupied 62.4 percent of the landscape revealed that 52.4 percent of the planning area was grassland with less than 10.0 percent big sagebrush canopy cover and 10.0 percent of the planning area was grasslands with greater than 10.0 percent big sagebrush canopy cover. Ponderosa pine and sparsely vegetated sites occupied approximately 2.8 percent and 10.8 percent of the planning area, respectively. Other land cover conditions occupied 17.6 percent of the planning area and was dominated by human disturbed areas (13.6 percent), riparian areas or water (2.8 percent), and areas that could not be classified (1.2 percent).

Vegetation and Ecological Site Sampling

The vegetation sampling conducted in 2003 to 2005 included 571 plots sampled in terrestrial ecosystems. In addition 18 plots were visited but rejected because they contained over 75 percent canopy cover by exotic species. Average annual precipitation during the years of this assessment was slightly above the long-term average (12.4 in; 31.5 cm) in 2003 (14.6 in; 37.1 cm) and in 2005 (13.4 in; 34.0 cm), however, the annual precipitation in 2004 was below average (9.9 in; 25.1 cm). Growing season precipitation (in other words, April through August) was below the long term average (8.5 in; 21.6 cm) in 2003 (7.6 in; 19.3 cm) and 2004 (5.1 in; 13.0 cm), but was slightly above average in 2005 (9.7 in; 24.6 cm).

Table 1—Omission/commission table for the assessment of land cover classification accuracy derived from SPOT 5 10-m multi-spectral imagery for the Thunder Basin planning area. Numbers in bold indicate the number of plots correctly identified, whereas non-bolded numbers in a given row indicate the number of plots omitted and the non-bolded numbers in a given column indicate the number of plots committed.

Omissions

		Land Cover Class							Classification Accuracy		
		Land Cover Class	Grassland	Sagebrush > 10%	Sparsely Vegetated	Disturbed	Ponderosa Pine	Water	Riparian	Omissions	Commissions
			-----Number of Plots -----								
Comissions	Grassland		20	11		3	2		1	71.4	54.1
	Sagebrush> 10%		8	22						66.7	73.3
	Sparsely Vegetated				7					100.0	100.0
	Disturbed					8				100.0	100.0
	Ponderosa Pine						28			93.3	100.0
	Water							26		100.0	100.0
	Riparian								0	0	100.0
Overall Classification Accuracy = 81.6											

Canopy Cover

Within each ecological site the mean canopy cover varied in each year we sampled (figure 2), which likely represents differences in the timing and quantity of moisture. Across ecological sites, the greatest canopy cover occurred in 2003 followed by 2005. Canopy cover was lowest in 2004 (figure 2), likely a result of the drought conditions that occurred throughout that growing season. The canopy cover variation we observed between 2003 and 2005 could be a function of ample spring moisture in 2003, which resulted in early plant growth during the growing season. This variation in canopy cover cannot be readily explained. Therefore, we used relative cover for analyses of plant community dominance to reduce the effects of annual variation in canopy cover.

Western wheatgrass was the most dominant species occurring across all of the terrestrial ecological sites followed by big sagebrush, needle and thread, blue grama, annual brome species and to a lesser extent threadleaf sedge and six weeks fescue (table 3). Annual brome, which includes cheatgrass (*Bromus tectorum*) and field brome (*Bromus arvensis*) were dominant species across several of the ecological sites we sampled, especially in saline upland sites. However, annual brome was not as prevalent on sandy and shallow sandy ecological sites. Other site preferences we observed included soapweed yucca and needle and thread preference for sandy sites, greasewood preference for saline upland, and big sagebrush preference for all ecological sites except for shallow sandy and saline upland.

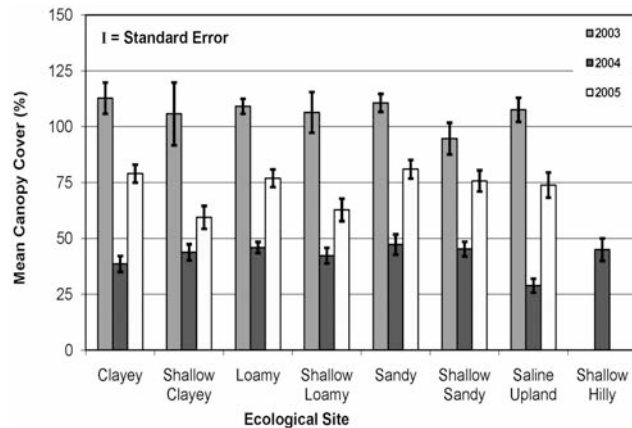


Figure 2—Mean understory canopy cover (+/- standard error of the mean) from 2003 through 2005 on terrestrial ecological sites in the Thunder Basin planning area.

Productivity

Productivity, measured as the dry weight of grasses and forbs differed varied across and within terrestrial ecological sites from 2003 through 2005 (figure 3). Drought conditions resulted in lower productivity readings across ecological sites in 2004 as compared to 2003 and 2005, and there was very little variation in productivity in 2004 across ecological sites. There was little variation in productivity between 2003 and 2005 on clayey, loamy, shallow loamy, or saline upland ecological sites. However, productivity was greater on shallow clayey, sandy, and shallow sandy ecological sites in 2003 as compared to 2005.

Vegetation Structure

Vegetation structure also varied across terrestrial ecological sites from 2003 through 2005 (figure 4). Drought conditions resulted in significantly lower vegetation structure readings across ecological sites in 2004 as compared to 2003 and 2005. Structure was uniformly low in 2004, with the exception of the shallow hilly ecological site, and showed little variation compared to 2003 and 2005. Vegetation structure was generally greater in 2005 than 2003, however, there was little variation in vegetation structure between these years in shallow loamy, shallow sandy and saline upland ecological sites. Differences in vegetation structure across ecological sites within a given year were most apparent in 2005, where clayey, loamy and sandy ecological sites had greater structure than shallow clayey, shallow loamy, shallow sandy, and saline upland ecological sites.

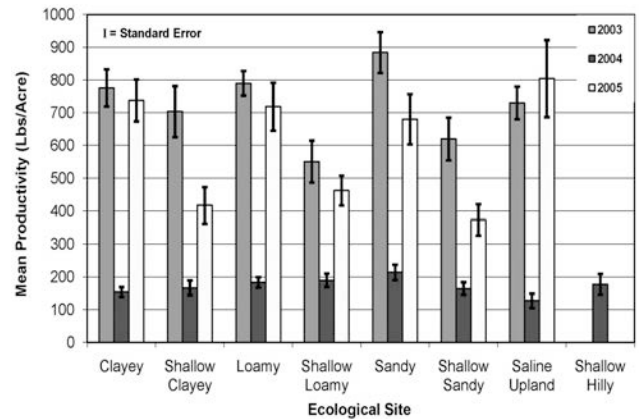


Figure 3—Productivity by ecological site (+/- standard error of the mean) from 2003 through 2005 on terrestrial ecological sites in the Thunder Basin planning area.

Exotic Species and Noxious Weeds

Exotic species were found in every ecological site we sampled. In fact, 57.4 percent of the plots sampled had greater than five percent relative cover of exotic plant species and 22.8 percent of the plots we sampled had over 20 percent relative cover of exotic species. Six of the exotic species sampled are also designated by the Wyoming Department of Agriculture as noxious weeds. The noxious weeds include nodding plumeless thistle (*Carduus nutans*), Canada thistle (*Cirsium arvense*), field bindweed (*Convolvulus arvensis*), gypsiflower (*Cynoglossum officinale*), leafy spurge (*Euphorbia esula*), field sowthistle (*Sonchus arvensis*), and saltcedar (*Tamarix* spp.). In addition we found one native species, skeletonleaf burr ragweed (*Ambrosia tomentosa*), which is also listed as a noxious weed by the Wyoming Department of Agriculture.

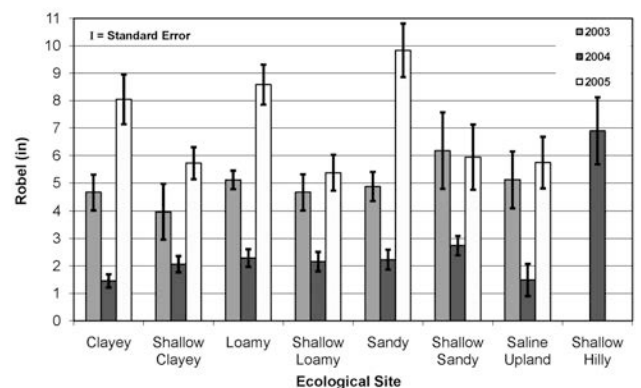


Figure 4—Vegetation structure measurements based on Robel pole readings (+/- standard error of the mean) from 2003 through 2005 on terrestrial ecological sites in the Thunder Basin planning area.

Table 2—Summary of acres within land cover classification categories identified with SPOT 5 satellite imagery for terrestrial ecological sites in the Thunder Basin region.

Ecological Site	SPOT 5 Satellite Imagery Land Cover Classification Categories				
	Grassland (ARTR2 ¹ <10%)	Grass & Shrubland (ARTR2 >10%)	Ponderosa Pine	Sparsely Vegetated	Other ²
-----acres-----					
Clayey	76,147	14,362	213	6,606	24,933
Shallow Clayey	118,883	15,518	351	51,811	35,620
Loamy	163,862	37,913	589	9,745	47,661
Shallow Loamy	39,419	5,331	299	6,041	10,890
Sands/Sandy	38,096	10,612	258	1,490	12,754
Shallow Sandy	21,651	4,186	87	4,270	4,228
Site Based Classification ²					
	Grass and Shrubland		Ponderosa Pine	Sparsely Vegetated	Other
Shallow Hilly	—		22,540	4,348	3,341
Very Shallow	1,810		2	189	787
Saline Upland	29,914		77	3,456	9,202
Badlands/ Gullied Lands	23,553		106	6,499	4,699

¹ Big sagebrush (*Artemisia tridentata*)² Other includes riparian, water, disturbed, and unclassified land cover classes.³ Site based classification indicates land cover classes that were classified based on topo-edaphic properties, independent of shrub cover.

The introduced species of greatest concern in the Thunder Basin planning area are cheatgrass and field brome, collectively referred to as annual brome (figure 5a, b). Annual brome occurred in 91 percent of the terrestrial plots we sampled and comprised a majority of the introduced species canopy cover. In most cases the relative proportion of annual brome did not differ within terrestrial ecological sites across the three years we sampled (figure 5a). However, differences do exist across some of the ecological sites, mainly a result of high proportions of annual brome on saline upland sites and relatively low proportions of annual brome on sandy and shallow sandy ecological sites. However, absolute cover of annual brome (figure 5b) was lower in 2004 in most of the ecological sites we sampled, and with the exception of sandy sites, annual brome cover did not vary between 2003 and 2005 on the other terrestrial ecological sites. The absolute cover patterns of annual brome essentially mirror the patterns we observed in total cover (figure 2). In general we found annual brome in fairly constant proportions across ecological sites from year to year, but annual brome responds favorably when the timing and quantity of precipitation is adequate and less favorably when precipitation is not adequate.

DISCUSSION

Grassland ecosystems of the Thunder Basin Region were historically dependent on grazing by native herbivores and fire as disturbance factors that shaped ecosystem diversity (Fuhlendorf and Engle 2001; Knapp and others 1999; Knight

1994). Historically, grazing by native herbivores, especially bison, played a significant role in shaping and maintaining terrestrial grass and shrub ecosystems (Hart and Hart 1997; Knapp and others 1999) and interacted with fire to create a shifting mosaic of conditions (Fuhlendorf and Engle 2001; Knapp and others 1999). Although grazing by domestic animals is currently the primary use of grass and shrub ecosystems, the foraging ecology of grazers that historically occupied the Great Plains differed from those used today (Plumb and Dodd 1993) and current grazing practices in grassland ecosystems have generally been found to dramatically differ from the historical role of herbivores (Fuhlendorf and Engle 2001). Existing livestock grazing practices have been focused on achieving even distribution of animals and even utilization, which produces relatively uniform or homogeneous vegetation conditions (Fuhlendorf and Engle 2004). Today, plant communities that have species compositions characteristic of light grazing regimes are nearly absent in terrestrial ecological sites within the Thunder Basin planning area. For example, species that typically decrease with increasing grazing pressure, like green needlegrass (*Nassella viridula*), bluebunch wheatgrass (*Pseudoroegneria spicata*), and Indian ricegrass (*Achnatherum hymenoides*) were rarely found in this landscape. Past and current grazing practices coupled with recent drought conditions have resulted in the current distribution of plant communities with plant species compositions indicative of moderate and heavy grazing regimes dominating terrestrial ecological sites, and plant communities with plant species compositions indicative of light grazing regimes almost totally lacking in the landscape.

Table 3—Dominant plant species in terrestrial ecological sites sampled in Thunder Basin, WY in from 2003 through 2005. Dominance is based on understory (vegetation <1m in height) quadrat sampling in each plot.

Plant Species	Relative Cover (%) of the 10 Most Dominant Plant Species by Ecological Site							
	Clayey	Shallow Clayey	Loamy	Shallow Loamy	Sandy	Shallow Sandy	Saline Upland	Shallow Hilly
Annual brome	7.9	8.3	9.6	7.9	6.5	3.4	15.1	2.1
Prairie sagewort	0.3	0.4	0.5	0.8	0.7	2.9	0.1	0.5
Purple threeawn	1.4	0.5	2.4	1.2	1.6	2.8	1.8	1.3
Big sagebrush	7.5	12.7	7.4	12.1	5.9	6.3	1.1	11.3
Blue grama	9.6	14.8	9.5	12.6	9.2	7.4	8.5	7.2
Needleleaf sedge	4.2	2.0	3.7	3.1	3.5	0.4	1.2	3.8
Threadleaf sedge	0.9	3.0	3.9	5.1	11.6	21.7	0.4	0.8
Slender wheatgrass	2.5	3.1	0.2	1.6	0.0	2.2	0.3	0.0
Needle and thread	3.4	3.8	6.3	7.7	11.6	13.4	1.1	6.0
Rocky Mtn. juniper	0.1	0.0	0.0	0.1	0.0	0.0	0.0	3.7
Prairie junegrass	1.5	2.1	1.1	2.4	1.9	2.0	0.1	2.2
Common pepperweed	1.6	0.6	0.9	0.4	0.5	0.4	2.6	0.0
Plains pricklypear	2.9	2.2	3.2	2.1	2.8	2.1	5.3	0.9
Western wheatgrass	29.0	19.3	26.8	18.2	23.4	5.8	25.4	9.1
Wooly plaintain	2.6	1.8	2.9	1.2	1.2	0.8	2.6	0.1
Bluegrass spp.	7.2	6.7	5.1	4.1	4.7	1.5	9.4	0.3
Bluebunch wheatgrass	0.2	1.3	0.0	1.7	0.0	1.2	0.0	11.6
Greasewood	0.2	0.2	0.0	0.2	0.0	0.0	3.9	0.0
Sixweeks fescue	4.1	1.8	6.3	4.2	4.5	2.6	3.8	0.3
Soapweed yucca	0.3	0.3	0.1	0.5	0.9	7.2	0.0	1.2
Average Absolute Cover (%)	76.8	69.7	77.4	70.5	79.6	71.9	70.1	40.0

Without fire as a disturbance process, many of these ecosystems would succeed to shrub or tree-dominated areas (Archer 1994). Despite a documented frequently occurring fire regime in the Thunder Basin region (Fisher and others 1987; Knight 1994; Perryman 1996; Perryman and Laycock 2000), public and private land management policy in the Thunder Basin planning area has encouraged and employed fire suppression programs in the surrounding grassland and shrubland areas since the 1940s (Perryman 1996). Although wildfires still occur within the landscape, and a few ranchers have tried an occasional prescribed burn, the extent of these fires and their scattered use have reduced the influence of this disturbance process in the landscape. The implications of the loss of this natural disturbance process on ecosystem diversity within Thunder Basin is poorly understood and not well documented. However, the role of fire throughout the Great Plains, with rare exceptions, has been eliminated which has modified species compositions and altered ecosystem function (Sieg and others 1999).

Introduced exotic plant species have been identified as one of the greatest threats to the integrity and productivity of native rangeland ecosystems and conservation of indigenous biodiversity (DiTomaso 2000; Mack and others 2000). In addition to environmental consequences, the damages caused and the costs incurred to control invasive plants are several billion dollars each year in wild and working landscapes of the United States (Pimental and others 2000). Although a number of exotic species occur in the Thunder Basin region, cheatgrass and field brome are the most dominant and frequently occurring species. Cheatgrass and field brome, are non-indigenous annual brome grasses that have invaded rangeland ecosystems of the western United States. Anthropogenic activities have been the primary contributors to the introduction and spread of annual brome (Gelbard and Belnap 2003; Mack 1981).

Invasion of rangeland ecosystems by annual brome has long been considered a point of contention in North America (Young and Allen 1997). Contention regarding annual brome stemmed from scientists and land managers who

disagreed about the potential impacts of annual brome invasion on native ecosystem structure and function and the potential of annual brome to provide a desirable forage resource for livestock (Young and Allen 1997). Annual brome is consumed by both livestock (Murray 1971; Haferkamp and others 2001a) and wildlife (Austin and others 1994), and at certain times of the year annual brome can provide high quality forage (Ganskopp and Bohnert 2001). However, annual brome has many attributes that make it an undesirable forage (Young and Allen 1997) especially its relatively short growing season (Ganskopp and Bohnert 2001) and its poor forage quality and palatability outside of the growing season. Annual bromes can substantially reduce the vigor of native plant communities through competitive reduction of water and nitrogen availability (Lowe and others 2003; Young and Evans 1978). Furthermore, controlling annual brome has been shown to result in several livestock production benefits including improving the forage quality of perennial native grass species (Haferkamp and others 2001b), producing higher livestock gains (Haferkamp and others 2001a), and converting degraded annual brome dominated areas back to native-dominated perennial grass communities (Evans and Young 1977).

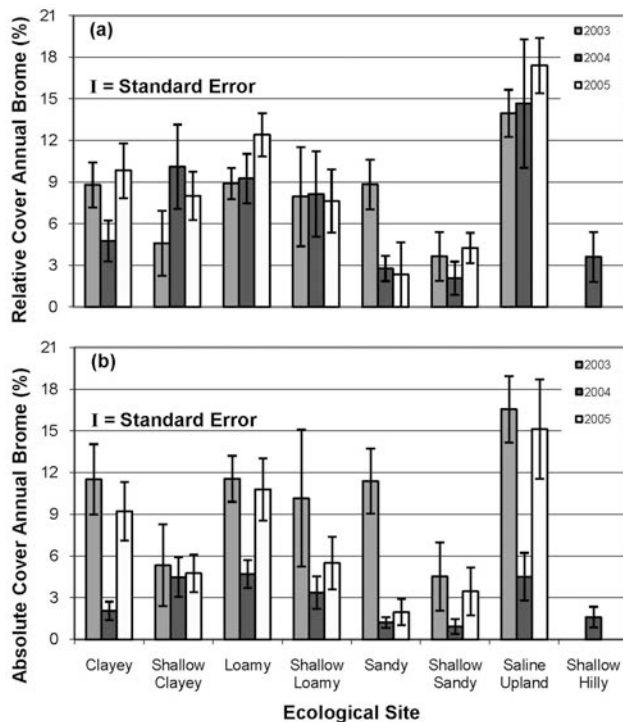


Figure 5—Relative cover $\pm$ standard error of the mean (a) and absolute cover $\pm$ standard error of the mean (b) of annual brome species (cheatgrass and field brome) from 2003 through 2005 on terrestrial ecological sites in the Thunder Basin planning area.

Annual brome invasion has been fueled by its competitive nature and has been associated with several negative environmental and economic consequences. Competitive attributes of annual brome include the ability to rapidly establish and attain community dominance following disturbances such as wildfire (Young and Evans 1978), the ability to rapidly respond to increases in resource availability (Lowe and others 2003; Norton and others 2004), and the ability to compete for water (Young and Evans 1977). Invasion by annual brome has resulted in alterations of rangeland ecosystem function and structure. These alterations include reductions in above and below ground biomass (Ogle 2003), increases in plant litter (Belnap and Phillips 2001), changes in plant community canopy architecture (Belnap and Phillips 2001), reductions in soil biota richness and abundance (Belnap and others 2005), reductions in plant community richness (Belnap and others 2005), and increases in wildfire frequency (Whisenant 1990).

MANAGEMENT IMPLICATIONS

Extractive activities associated with the rich natural resources of this area (for example coal bed methane, coal, and oil) have resulted in direct land conversions of some of the Thunder Basin area, but the area has not seen extensive direct conversion to other land uses. Despite the lack of direct conversion, grass and shrub ecosystems in the Thunder Basin planning area appear to have been influenced by fire regime modification, modification of historical grazing regimes, and the spread of introduced species. Conservation planning efforts aimed at improving ecosystem diversity and wildlife habitat in this working landscape should strategically target degraded ecosystems with modest re-introductions of prescribed fire, chemical control of the annual brome species, augmentative seeding to re-establish desirable grass and forb species and improve species diversity, and grazing modifications designed to promote healthy plant communities and wildlife habitat for species of concern. This assessment has identified ecosystems in need of improvement and could be further integrated with wildlife habitat objectives for this planning area to identify the specific locations best suited for ecosystem improvement treatments. For the Thunder Basin region to remain an ecologically significant grassland, greater attention must be paid to retaining and restoring functional native ecosystem diversity.

ACKNOWLEDGMENTS

Funding for this project was provided to the Thunder Basin Grasslands Prairie Ecosystem Association through grants from the U.S. Fish and Wildlife Service, the Natural Resource Conservation Service, Powder River Coal

Company, the National Fish and Wildlife Foundation, the Bureau of Land Management, the U.S. Forest Service, the Leopold Stewardship Fund of the Sand County Foundation, the Sonoran Institute, the Wyoming Governor's Federal Policy Fund, Rio Tinto Energy America (previously Kennecott Energy Company), Converse and Weston Counties of Wyoming, and a grant to the Ecosystem Management Research Institute from the Bradley Fund for the Environment. We appreciate the support and assistance given by the private landowners in this region, State and Federal agency personnel, seasonal field assistants, and the many people who contributed ideas and knowledge to this project.

REFERENCES

- Archer, S. 1994. Woody plant encroachment into southwestern grasslands and savannas: rates, patterns and proximate causes. Pages 13-68 in Vavra, M., Laycock, W., Pieper, R., eds. Ecological implications of livestock herbivory in the West. Society for Range Management, Denver, Colorado, USA.
- Austin, D.D., Stevens, R., Jorgensen, K.R., Urness, P.J. 1994. Preferences of mule deer for 16 grasses found on Intermountain winter ranges. *Journal of Range Management* 47:308-311.
- Belnap, J., Phillips, S.L. 2001. Soil biota in an ungrazed grassland: response to annual grass (*Bromus tectorum*) invasion. *Ecological Applications* 11:1261-1275.
- Belnap, J., Phillips, S.L., Sherrod, S.K., Moldenke, A. 2005. Soil biota can change after exotic plant invasion: does this affect ecosystem processes? *Ecology* 86:3007-3017.
- Canfield, R.H. 1941. Application of the line interception method in sampling range vegetation. *Journal of Forestry* 39:388-394.
- DiTomaso, J.M. 2000. Invasive weeds in rangelands: species, impacts, and management. *Weed Science* 48:255-265.
- Evans, R.A., Young, J.A. 1977. Weed control-revegetation systems for big-sagebrush-downy brome rangelands. *Journal of Range Management* 30:331-336.
- Fisher, R.F., Jenkins, M.J., Fisher, W.F. 1987. Fire and the prairie-forest mosaic of Devils Tower National Monument. *American Midland Naturalist* 117:250-257.
- Forrest, S.C., Strand, H., Haskins, W.H., Freese, C., Proctor, J.D., Dinerstein, E. 2004. Ocean of grass: a conservation assessment for the Northern Great Plains. Northern Plains Conservation Network and Northern Great Plains Ecoregion, World Wildlife Fund, Bozeman, MT. Pages 18.
- Fuhlendorf, S.D., Engle, D.M. 2001. Restoring heterogeneity on rangelands: ecosystem management based on evolutionary grazing patterns. *Bioscience* 51:625-632.
- Fuhlendorf, S.D., Engle, D.M. 2004. Application of the fire-grazing interaction to restore a shifting mosaic on tallgrass prairie. *Journal of Applied Ecology* 41:604-614.
- Ganguli, A.C., Haufler, J.B., Mehl, C.A., Chew, J.D. 2008. Incorporating historical ecosystem diversity into conservation planning efforts in grass and shrub ecosystems. Abstracts, Fifteenth Wildland Shrub Symposium, Wildlands and Wildlife Habitat, 1998 June 17-19, Bozeman, MT.
- Ganskopp, D., Bohnert, D. 2001. Nutritional dynamics of 7 northern Great Basin grasses. *Journal of Range Management* 54:640-647.
- Gelbard, J.L., Belnap, J. 2003. Roads as conduits for exotic plant invasions in a semiarid landscape. *Conservation Biology* 17:420-432.
- Haferkamp, M.R., Grings, E.E., Heitschmidt, R.K., MacNeil, M.D., Karl, M.G. 2001a. Suppression of annual brome impact rangeland: animal responses. *Journal of Range Management* 54:663-668.
- Haferkamp, M.R., Heitschmidt, R.K., Grings, E.E., MacNeil, M.D., Karl, M.G. 2001b. Suppression of annual brome impacts rangeland: vegetation responses. *Journal of Range Management* 54:656-662.
- Hart, R.H., Hart, J.A. 1997. Rangelands of the Great Plains before European settlement. *Rangelands* 19:4-11.
- Knapp, A.K., Blair, J.M., Briggs, J.M., Collins, S.L., Hartnett, D.C., Johnson, L.C., Towne, E.G. 1999. The keystone role of bison in North American tallgrass prairie. *Bioscience* 49:39-50.
- Knight, D.H. 1994. Mountains and Plains: The Ecology of Wyoming Landscapes. Yale University Press, New Haven, CT.
- Lowe, P.N., Laurenroth, W.K., Burke, I.C. 2003. Effects of nitrogen availability on competition between *Bromus tectorum* and *Bouteloua gracilis*. *Plant Ecology* 167:247-254.
- Mack, R.N. 1981. Invasion of *Bromus tectorum* L. into western North America: An ecological chronicle. *Agro-Ecosystems* 7:145-165.
- Mack, R.N., Simberloff, D., Lonsdale, W.M., Evans, H., Clout, M., Bazzaz, F.A. 2000. Biotic invasions: causes, epidemiology, global consequences, bunchgrass ranges in southern Idaho. *Journal of Range Management* 24:407-410.
- Murray, R.B. 1971. Grazing capacity, sheep gains: cheatgrass, bunchgrass ranges in southern Idaho. *Journal of Range Management* 24:407-410.
- Norton, J.B., Monaco, T.A., Norton, J.M., Johnson, D.A., Jones, T.A. 2004. Soil morphology and organic matter dynamics under cheatgrass and sagebrush-steppe plant communities. *Journal of Arid Environments* 57:445-466.
- Ogle, S.M., Reiners, W.A., Gerow, K.G. 2003. Impacts of exotic annual brome grasses on ecosystem properties of Northern Mixed Grass prairie. *American Midland Naturalist* 149:46-58.
- Pechanec, J.F., Pickford, G.D. 1937. A weight-estimate method for the determination range pasture production. *Agronomy Journal* 29:894-904.
- Perryman, B.L. 1996. The vegetation mosaic of the Rochelle Hills area of the Thunder Basin National Grasslands. In: Fire history of the Rochelle Hills area of the Thunder Basin National Grasslands. PhD Dissertation, University of Wyoming, Laramie.
- Perryman, B.L., Laycock, W.A. 2000. Fire history of the Rochelle Hills Thunder Basin National Grasslands. *Journal of Range Management* 53:660-665.
- Pimentel, D., Lach, L., Zuniga, R., Morrison, D. 2000. Environmental and economic costs of nonindigenous species in the United States. *Bioscience* 50:53-65.
- Plumb, G.E., Dodd, J.L. 1993. Foraging ecology of bison and cattle on a mixed prairie: implications for natural area management. *Ecological Applications* 3:631-643.
- Robel, R.J., Briggs, J.N., Dayton, A.D., Hulbert L.C. 1970. Relationship between visual obstruction measurements and the weight of grassland vegetation. *Journal of Range Management* 23:295-297.
- Sieg, C.H., Flather, C.H., McCanny, S. 1999. Recent biodiversity patterns in the Great Plains: implications for restoration and management. *Great Plains Research* 9:277-313.
- USDA, NRCS. 2003. National range and pasture hand book. Washington, D.C.
- USDA, NRCS. 2006. The PLANTS Database, Version 3.5 (<http://plants.usda.gov>). Data compiled from various sources by Mark W. Skinner. National Plant Data Center, Baton Rouge, LA.
- Whisenant, S.G. 1990. Changing fire frequencies on Idaho's Snake River Plains: ecological and management implications. In: McArthur, E. D., Romney, E.M., Smith, S.D., Tueller, P.T. eds. Proceedings of the Symposium on cheatgrass invasion, shrub die-off, and other aspects of shrub biology and management. p.4-10. USFS-INT-GTR-313.
- Young, J.A., Allen F. L. 1997. Cheatgrass and range science: 1930-1950. *Journal of Range Management* 50:530-535.
- Young, J.A., Evans, R.A. 1978. Population dynamics after wildfires in sagebrush grasslands. *Journal of Rangeland Management* 31:283-289.

Comparing Vegetation Monitoring Methods in Shrublands: How Valuable is Grant's Method in Shrub Communities?

Matt M. Church¹, Mary I. Williams¹, Ann L. Hild², and Ginger B. Paige¹

ABSTRACT

Faced with managing extensive rangelands, land managers need reliable monitoring methods. Grant and others (2004) proposed a monitoring method that uses a floristic classification of dominant vegetation and assesses species frequency. The method was developed in native grass prairies with limited woody vegetation and is designed to provide more efficient repetitive monitoring. Our objective for this study was to determine if Grant's method is useful in shrubland systems for a newly implemented baseline inventory and monitoring program. We conducted a study to compare Grant's method and its efficacy to more commonly used line-point intercept in grass and shrub alliances at Brown's Park National Wildlife Refuge, Colorado. In the summer of 2007 we conducted baseline inventory of vegetation characteristics on 39 permanent bottomland plots in six pre-determined grass and shrub vegetation alliances on the refuge. Within each plot, we monitored vegetation with line-point intercept and Grant's method on three 50 m line transects. We compare data collected using these methods from a grassland (smooth brome dominated) and a shrubland (greasewood dominated) alliance. Results of this study indicate that data collected using Grant's method is more variable than line-point intercept data in shrub systems. Bare ground was detected less using Grant's than line-point intercept; however, Grant's detected more herbaceous and invasive species overall than did the line-point intercept method. A complementary use of the methods for long-term monitoring is recommended that accommodates tradeoffs between incorporating detail versus efficiency of data collection efforts.

INTRODUCTION

Rangeland inventory and long-term monitoring allow land managers to detect changes and assess trends in rangeland resources. The initial data collection on a site is to develop a baseline inventory; subsequent monitoring is valuable for detecting changes and assessing trends over time. Baseline inventory provides both ecological and management information about the site. Effective monitoring practices enable managers to evaluate the effectiveness of their management actions and develop more appropriate management practices over time (Barker 2001).

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Matt M. Church, Graduate Research Assistant, Mary I. Williams, Doctoral Research Assistant and Ginger B. Paige, Assistant Professor, Department of Renewable Resources, University of Wyoming, Laramie, WY

²Ann L. Hild, Professor, Renewable Resources Dept. University of Wyoming, Laramie. annhild@uwyo.edu

There are a variety of vegetation and site characteristics that can be monitored and the methods differ in rigor and resource commitments. Furthermore, vegetation monitoring methods vary greatly depending on location, scale, and objectives. Plant species composition may be quantified using techniques that assess frequency of occurrence, cover, or density (Elzinga and others 1998). Herrick and others (2005a and 2005b) describe many of the commonly used rangeland monitoring methods. These include line point intercept and belt transects which assess percent cover and density of vegetation species. Grant and others (2004) proposed a monitoring method that uses a floristic classification of dominant vegetation and assesses their frequency. The method was developed in native grass prairies with limited woody vegetation and is designed to provide for more efficient repetitive monitoring on the same site.

Land managers, faced with managing and monitoring large areas of rangelands, are interested in time efficient but effective monitoring methods. We conducted a study to compare Grant's method and its efficacy to more commonly used monitoring methods, specifically the line point intercept method. The overall objective was to determine if Grant's method is useful in shrubland systems in a newly implemented baseline inventory and monitoring program. The methods are compared in both shrub and grasslands systems.

METHODS

Area of Study

The study was conducted in the bottomlands area of Brown's Park National Wildlife Refuge (BPNRW) in northwestern Colorado. BPNRW is located in the Upper Colorado River Ecosystem and includes a mixture of habitats from the Middle Rocky Mountains, Wyoming Basin, and Colorado Plateau (USFWS 1999). The refuge is comprised of 5445 ha of wetlands, riparian habitats, and uplands and provides habitat for 300 wildlife species including 68 mammals. This region is semi-arid, and receives approximately 24 cm of precipitation a year. Elevations at BPNRW vary from 1,632 to 1,889 m.

The bottomlands at BPNRW are dominated by dense stands of greasewood (*Sarcobatus vermiculatus*), rabbitbrush (*Chrysothamnus* spp.), perennial grasses and several

invasive plant species. Portions of the bottomlands are periodically flooded to provide habitat for migratory birds. Shrublands account for approximately 660 ha of the 1150 ha bottomland area. The refuge is currently implementing new baseline monitoring programs. Baseline inventory and long-term monitoring of the bottomlands are needed for the refuge to develop and assess their management programs.

Plot Layout

Thirty-nine to 55 m fixed radius circular plots were located on identified bottomland alliances on the refuge, spaced at least 250 m apart (figure 1). The center of each plot was marked and its location recorded with a global positioning system (GPS). Soil and vegetation characteristics were measured on three 50 m line transects arranged in a spoke

design originating 5 m from the center of the plot. An azimuth direction (1° to 360°) for the first transect was chosen from a random number table and subsequent transects were placed at 120° intervals from the first transect. The six alliances monitored in the bottomlands at BPNWR include: alkali sacaton (*Sporobolus airoidies*) and inland saltgrass (*Distichlis spicata*) 5 plots, needlegrass (*Stipa* spp) -greasewood mix, 2 plots, wheatgrass mix (*Elymus* spp) 6 plots, rabbitbrush mix, 10 plots, bottomland black greasewood dominated areas, 13 plots and smooth brome (*Bromus inermis*) 3 plots. For this study we concentrated on the plots in the smooth brome and bottomland greasewood alliances.

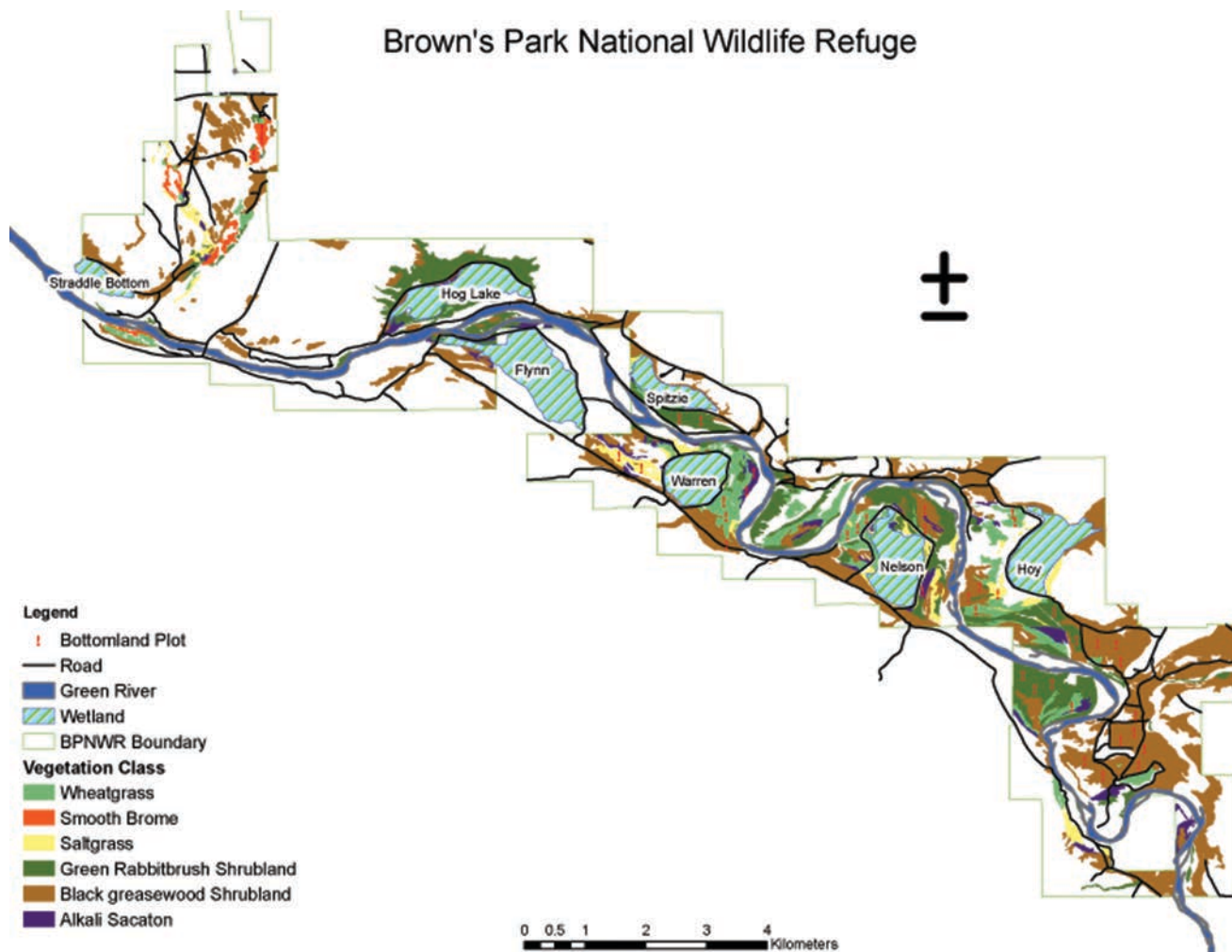


Figure 1—Bottomland study plots (39 to 100 m radius) were established within six vegetative alliances in bottomlands at Brown's Park National Wildlife Refuge, CO to initiate long-term monitoring. Monitoring data from the Smooth brome and greasewood dominated plots are used to compare monitoring methods.

Monitoring Methods

We monitored vegetation cover using both line-point intercept and Grant's methods on 3 plots in the Bottomland Greasewood alliance and 3 plots in the Smooth Brome alliance. Grant's method measured floristic characteristics on each 50 m line transect in a 0.1 x 0.5 m belt. A plant species was documented if it had greater than 10 percent abundance in each quadrat. Vegetation was recorded as frequency of occurrence and more than one species could be recorded in a quadrat. When no species had 10 percent or greater abundance within a quadrat, we recorded the quadrat as sparsely vegetated. More detailed description of the method is available (Grant and others 2004).

We used a modification of the line-point intercept (Canfield 1941) described by Herrick and others (2005b) to record vertical density measurements and total canopy and ground cover. Starting at the half-meter mark, and continuing every meter for 50 total points, we recorded cover data along each of the three 50 m transect lines within each plot. The first plant species intercepted (by a vertically dropped pin flag) and its height was recorded as top canopy; additional species at the same point were recorded as lower canopy in order of interception. A canopy species was recorded only once per point even if intercepted at more than one level. Ground surface was recorded as moss, rock (> 5 mm diameter), plant species base, litter or bare ground.

Method Comparison

Data from Grant's method are expressed as frequency of species occurrence, whereas line-point intercept data are expressed as percent cover. The data from the two methods cannot be compared directly because of these inherent differences. Grant data are continuous, with 100 quadrats along each line transect. Line-point intercept data are discontinuous with 50 distinct measurement points along each line transect. We recalculated the Grant data for a 0.1 x 1.0 m quadrat size. The recalculated counts for species were multiplied by two to result in percent occurrence. Recalculation of Grant transect data allowed for a comparable scale with the line-point intercept data which were collected at 1 m increments. We examined cover data of dominant shrub and grass species and bare ground from the shrubland and grassland alliances to compare the two methods. To determine the reliability of each measurement method within the two systems, we calculated the coefficients of variation (CV) for each plot. The CVs were analyzed to compare the variability of the two methods using analysis of variance (ANOVA).

RESULTS

Compared over all species, the CVs of line point intercept data were significantly less than the CVs of Grant measures

($P < 0.05$). More herbaceous species were detected by Grant's method in smooth brome and greasewood alliances (table 1). Bare ground was not as readily detected using Grant's method. Line-point intercept detected more abundant species but did not intercept less common species on the transects. Grant's method detected both abundant and less common species (such as VUOC, ELTR).

In the smooth brome alliance, the methods differ in their ability to detect bare ground under canopy cover (figure 2). The line-point intercept detected a small amount of bare ground in two of three plots; no bare ground was detected using Grant's method in the same plots. Smooth brome cover was recorded in each of the three plots using all methods; the amount detected was highly variable within and among the three plots regardless of the method used. Grant's detected less smooth brome with less variability at the 0.5 m than the 1.0 m scale. Detection of invasive species (CIAR, CIVU, IVAX) was consistently less using the line-point method than with Grant's. Overall, species detection within the three plots varied greatly even within one method, and there were no consistent trends among the three methods.

In the greasewood alliance, each method detected bare ground, however, the amount detected varied among methods (figure 3). Additionally, bare ground was quite different among the three plots when using the same method, suggesting that the characteristics of each plot differ and that plot differences are recorded by these methods. The three methods were not consistent in the amount of variability included in recording bare ground. Grant 1.0 m scale consistently detected more greasewood, with less variability than the line-point intercept method. The ability to detect the invasive cheatgrass (*Bromus tectorum*, BRTE) differed among the methods. In two of the plots, cheatgrass data were more variable using Grant's method than the line-point intercept. In the third plot however, Grant's detected cheatgrass on only one of three transects.

DISCUSSION and CONCLUSION

Differences in species and bare ground detection may be attributed to the fundamental differences inherent in each method. Grant's method may not be as time-efficient as the line-point intercept in shrub systems because it requires a lot of time *a priori* to identify alliances presently on the landscape; much of the labor and time is completed before actually getting to the field. Additionally, Grant's method uses an ocular estimation of species within quadrats, whereas the line-point intercept uses interception at and below the canopy level to detect species and ground cover. Bare ground was detected best in smooth brome vegetation

using the line-point intercept because the line point method forces observation of soil surface. This same issue was not evident in the greasewood where bare ground was more visible. The precision of the line-point intercept appears to focus observations and reduce variability within datasets. Ocular estimation of Grant's method is easy to use, but much more subjective. Grant's method, using the smaller quadrat (0.5 m) seems to have introduced excessive variation in the data when shrubs are present.

To take full advantage of the different strengths and minimize observed limitations, the two methods could best be used in combination in a long-term monitoring program. The addition of Grant's method may increase observation of species with minimal cover that are not readily detected by line-point intercept (in other words indicator or threatened species; Herrick and others 2005b). However, used alone, Grant's method may not accurately depict the coverage of bare ground and may introduce variation that

makes it difficult to discern management trend. For this reason, if the focus of the study is species presence and floristics, Grant's may be preferred over line-point intercept. However, because site characteristics such as bare ground are critical to ecosystem function, and because managers must document response to management actions over time (trend), use of only the Grant method may not be effective.

Given the ease of use of Grant's method, it could be incorporated with almost any other monitoring program. Because Brown's Park National Wildlife Refuge requires a rigorous baseline vegetation monitoring program, Grant's method could be incorporated into other monitoring methods as a frequent repeat monitoring technique, once more detailed baseline inventories are complete. We suggest using Grant's method for repeat interim monitoring, coupled with more detailed, less frequent monitoring methods such as line-point intercept and belt transects.

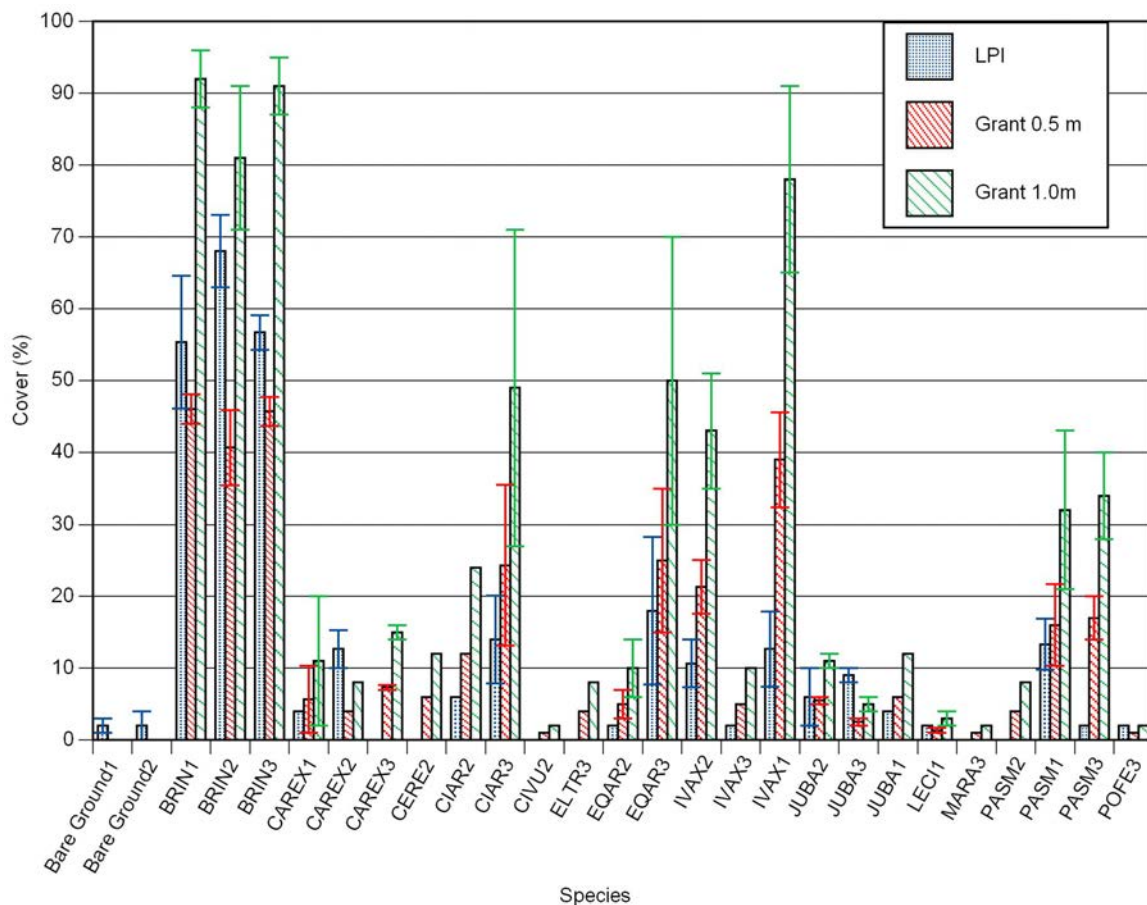


Figure 2—Cover attributes of three smooth brome plots recorded using three monitoring methods (line-point intercept (LPI), Grant 0.5 m resolution and Grant 1.0 m resolution). Error bars are one standard error of the mean per plot (3 line transects.) Species columns without error bars were recorded on only one of three transects within a plot. Species lack columns for a given method when the species was not detected by the method. Species name abbreviations are detailed in table 1.

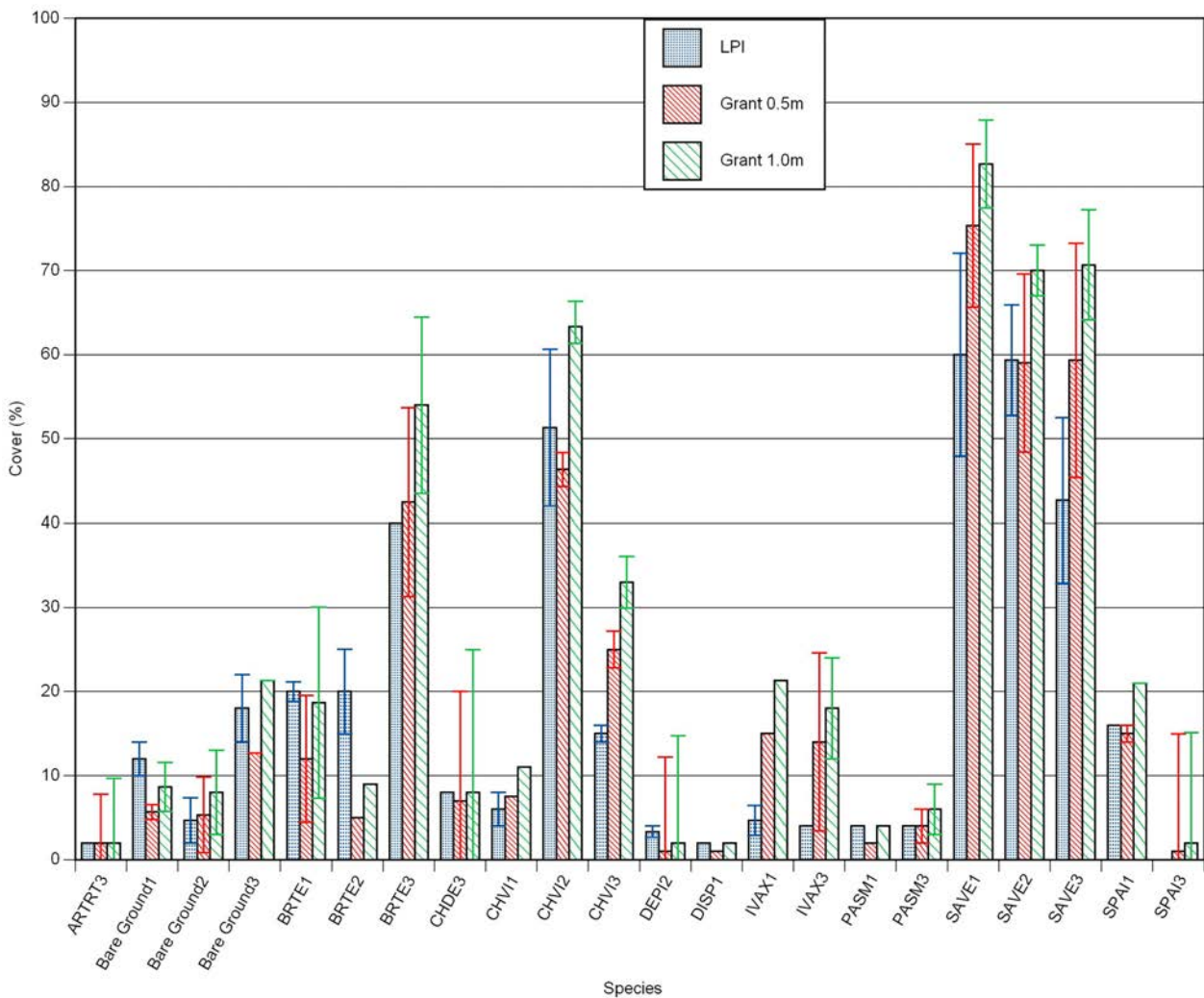


Figure 3—Cover attributes of three greasewood plots recorded using three monitoring methods (line-point intercept (LPI), Grant 0.5 m resolution and Grant 1.0 m resolution). Error bars are one standard error of the mean per plot (3 line transects). Species columns without error bars were recorded on only one of three transects within a plot. Species lack columns for a given method when the species was not detected by the method. Species name abbreviations are detailed in table 1.

REFERENCES

- Barker, P. 2001. A technical manual for vegetation monitoring. Resource Management and Conservation, Department of Primary Industries, Water and Environment. Hobart, Tasmania. 79 p.
- Canfield, R.H. 1941. Application of the line intercept method in sampling range vegetation. *Journal of Forestry*. 39:388-394.
- Elzinga, C.L., Salzier, D.W., Willoughby, J.W. 1998. Measuring & monitoring plant populations. BLM Technical Reference 1730-1. 477 p.
- Grant, T.A., Madden E.M., Murphy, R.K., Smith, K.A., Nenneman, M.P. 2004. Monitoring native prairie vegetation: the belt transect method. *Ecological Restoration*. 22:106-112.
- Herrick, J.E., Van Zee, J.W., Havstad, K.M., Burkett, L.M., Whitford, W.G. 2005b. Monitoring manual for grassland, shrubland, and savanna ecosystems. Volume II: Design, supplementary methods and interpretation. USDA-ARS Jornada Experimental Range, Las Cruces, NM. 200 p. http://usda-ars.nmsu.edu/monit_assess/monitoring.php
- U.S. Fish and Wildlife Service. 1999. Brown's park national wildlife refuge comprehensive conservation plan. 83 p.

A Metric System for Evaluating Off-site Mitigation for Ecosystem Services and Wildlife Habitat in Sagebrush Ecosystems

Jonathan B. Haufler¹ and Lowell H. Suring²

ABSTRACT

A fundamental concept of mitigation is that it is possible to compensate for impacts to ecosystem services at one site (the impact site) by replacing or increasing the same services at another site (the mitigation site). A challenge in the use of off-site mitigation is assuring that ecosystem services, including wildlife habitat, produced by off-site mitigation are commensurate with on-site impacts. Recent increases in energy developments within the sagebrush biome have raised concerns about impacts associated with these activities and efforts to mitigate those impacts. To help address these concerns, we developed a metric system to quantify impact losses and mitigation benefits based on a combination of NRCS Ecological Sites, existing vegetation conditions, and habitat assessment conducted at the landscape level for sagebrush-associated wildlife species. Changes to vegetation conditions within sagebrush ecosystems produced by on-the-ground mitigation treatments or by impacts are quantified based on comparison to a reference standard developed from the Ecological Site Description for the specific plant communities associated with either the mitigation or impact site. Wildlife benefits are also evaluated at a landscape scale using models that quantify the gains or losses in habitat quality associated with the mitigation or development activities. This metric system provides a standardized way of quantifying gains and losses of ecosystem services and wildlife habitat associated with impacts and mitigation which will help to ensure that gains associated with mitigation activities are commensurate with losses resulting from development.

INTRODUCTION

A fundamental concept for mitigation is that it is possible to compensate for impacts to ecosystem services at one site (the impact site) by replacing or increasing the same services at another site (the mitigation site). A challenge in the use of off-site mitigation is assuring that ecosystem services, including wildlife habitat, produced by off-site mitigation are commensurate with on-site impacts. A related challenge is to be able to effectively quantify the losses and gains in services in a scientifically valid and reproducible manner.

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.

¹Jonathan B. Haufler, Executive Director, Ecosystem Management Research Institute, Seeley Lake, MT. Jon_Haufler@emri.org

²Lowell H. Suring, Wildlife Ecologist, USDA Forest Service, Terrestrial Wildlife Ecology Unit, Aquatic Sciences Laboratory, Boise, Idaho. lsuring@fs.fed.us

Recent increases in energy developments within the sagebrush biome have raised concerns about impacts associated with these developments and efforts to mitigate these impacts. Substantial interest exists in use of off-site mitigation if this can be shown to produce benefits that are equivalent to the impacts. This paper describes a framework and measurements for mitigation metrics that can evaluate and quantify impacts to ecosystem services at development sites and compare and quantify these with commensurate amounts of ecosystem services produced through off-site mitigation.

THE CONCEPTUAL UNDERPINNINGS

It should be noted that it is not assumed that all impacts on all sites can be compensated through off-site mitigation. Some sites of high ecological value should be protected from development activities because their value to ecosystem services such as wildlife habitat that cannot be effectively mitigated and therefore should not be impacted. The methodologies described here are directed at those sites where development has been approved as an appropriate activity and therefore seek to mitigate on-site impacts through use of off-site mitigation.

The basic framework for quantifying services lost at impact sites and gained at mitigation sites requires assessments of the following:

1. The *existing level* of services provided at the impact and mitigation sites prior to the initiation of development and mitigation activities. These services are considered to be influenced by both the characteristics of the specific impact and mitigation sites as well as the surrounding landscape;
2. The *resulting level* of services expected at the impact and mitigation sites after the impact development and mitigation activities are carried out, considering both site and landscape effects;
3. The *duration* of the change (or period of time over which a change in services occurs) at the impact and mitigation sites; and
4. The *length of time* before the mitigation is expected to be fully successful at the mitigation site.

By assessing the above factors, it should be possible to design a metric system that allows consistent quantification of mitigation required to offset expected impacts from a

broad range of development activities. The basic units proposed to quantify benefits associated with mitigation activities or detriments associated with development impacts are really a variety of ecosystem services lost or gained over time. They are calculated in the same manner, so that a “credit unit” has an equal, but opposite, value as a “debit unit.” Thus, the benefits of credit units produced are intended to fully and specifically offset the detrimental debit units from a development.

Considerations Associated with Mitigation Metrics For Sagebrush Ecosystems

In the case of the sagebrush biome, the number of credit units or debit units associated with any activity should be a function of the following factors:

1. The area affected by the activity;
2. The ecological sites occurring in the affected areas;
3. The existing conditions within the area (essentially a measure of quality evaluated relative to a baseline);
4. The extent of change (positive or negative) caused by the activity relative to the existing conditions;
5. The spatial or landscape context in which the area is located (related primarily to habitat quality for selected species); and
6. The timing and duration of the expected change.

The discussion that follows elaborates on these factors. What is presented here is a general framework of factors that should be included in the determination of credit units and debit units. An example is provided to show how the metrics framework could be applied. Its actual utilization would need to be specifically tailored and calculated for a location.

Area Affected

The quantification of credit or debit units begins with a determination of the size and characteristics of the area affected by any action. Development activities will often affect both the specific area directly impacted by the development (the physical “footprint” of the activity), and additional area where the activity is expected to exert a negative influence (for example, effects of noise, disturbance, visual features, or other impacts that discourage use of nearby areas by sagebrush-associated wildlife). The additional areas impacted by a disturbance beyond the actual development footprint are considered in the landscape context, discussed below. Basic units for impacts or mitigation are the size (in ac or ha) of the actual footprint of the impact or mitigation site. This area will then be modified by the level of change in ecosystem services as a result of the development or mitigation treatments. In many cases, development will create a change from existing conditions and the level of ecosystem services that they

currently provide to no provision of ecosystem services with the development activity. For mitigation sites, the area treated will have an existing value of ecosystem services that should be increased through mitigation treatments. If a site is improved by 50 percent, then the number of ac mitigated would be multiplied by this level of improvement to determine the number of mitigation credit units produced.

Ecological Sites

The area affected must be characterized in terms of its existing and inherent (potential) conditions. Natural Resources Conservation Service (NRCS) ecological sites (<http://esis.sc.egov.usda.gov/>) provide a classification system that can facilitate identification of biotic and underlying abiotic drivers of ecosystem diversity that could provide consistency for measuring ecosystem services and thus mitigation benefits. Ecological sites classify areas that have similar soils and other abiotic and biotic conditions for defined precipitation zones within a Major Land Resource Area. Major Land Resource Areas (MLRA) are geoclimatically defined areas delineated by NRCS that have been mapped for the entire U.S. (NRCS 2006, <http://soils.usda.gov/survey/geography/mlra/>). Ecological site classifications have been developed for most MLRA's, with ecological site descriptions developed for each specific ecological site within these MLRA's. These sites are linked to soils, and are therefore mapped wherever NRCS soils mapping has occurred.

For each ecological site, various plant communities described as specific “states” as influenced by natural or anthropogenic disturbances have been identified. The dynamics of these plant communities or states are incorporated into a state and transition model for each site. Changes among states are defined as “transitions,” with some changes crossing “thresholds” that may make transition back to a prior state difficult (Friedel 1991; Laycock 1991). Various states that might occur on each ecological site have been described in ecological site descriptions (ESDs) for most MLRAs in the Rocky Mountain West, with work proceeding on those areas not yet completed. Descriptions of states for a specific ecological site should include all of the states that occurred historically under historical disturbance regimes (historical states), and other states produced as a result of recent (post-European settlement) anthropogenic influences including introduction of exotic species (anthropogenic states). Past influences of Native Americans are incorporated as part of the historical states. Some ecological site descriptions have not included descriptions of the full range of historical states and transitions, so these may need further development for some MLRA's. A full state and transition model for an ecological site should include descriptions of all of the states that occurred historically as well as any currently common states produced by anthropogenic influences (for example, see figure 1).

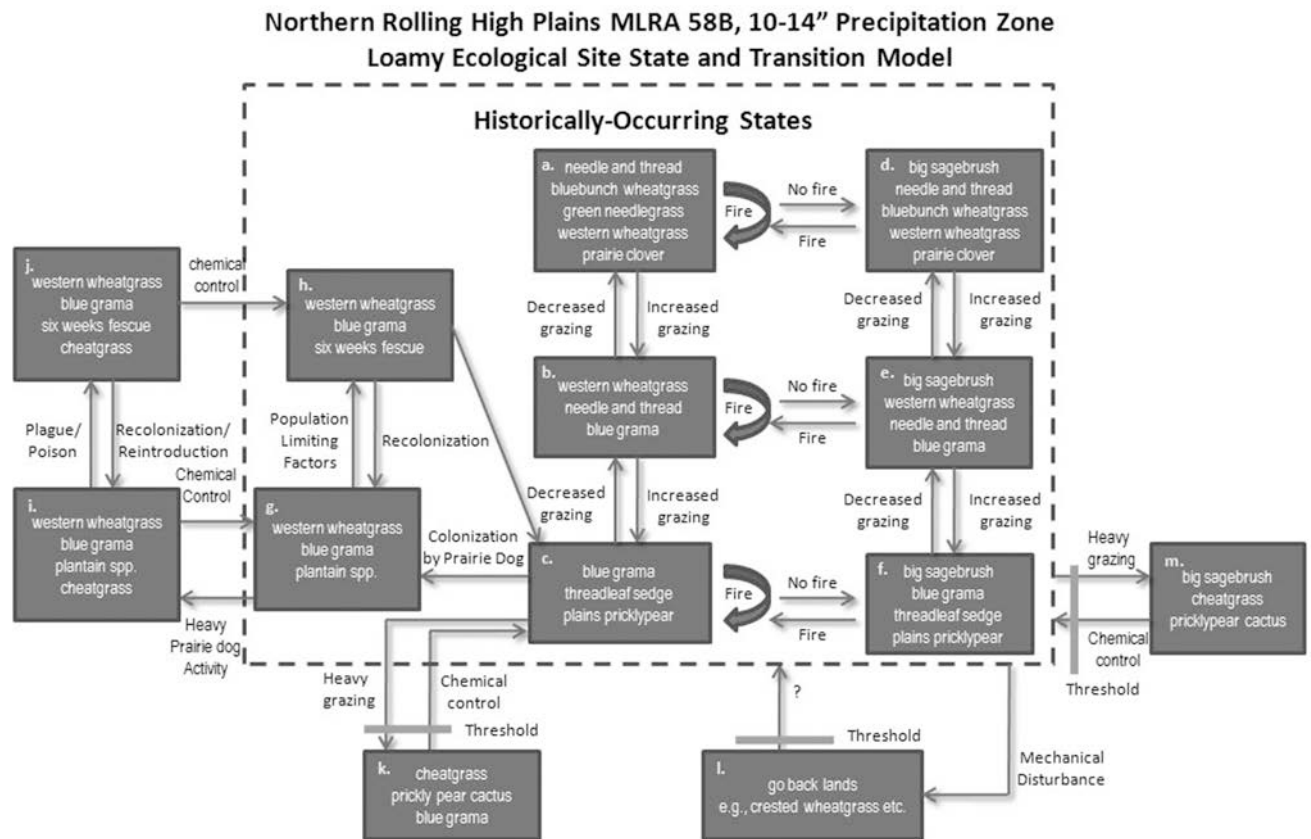


Figure 1—State and transition model for loamy ecological sites within the 10-14" precipitation zone of MLRA 58B, the Northern Rolling High Plains. States identified outside of the dashed box did not occur historically but rather are the result of recent anthropogenic changes.

Use of ecological sites as defined by NRCS assures that ecosystem services are being considered in equivalent locations in terms of having a similar abiotic environment. For example, two loamy ecological sites within the same MLRA and precipitation zone should have the potential of supporting similar states with similar potential productivity and thus have the potential to contribute similar ecosystem services. The services they are producing at any time will be determined by the existing plant community occurring at that time, but the potential of loamy sites should basically be the same.

A saline upland ecological site in the same MLRA and precipitation zone would have different plant communities or states associated with it than the loamy ecological site, as the different soil properties favor the occurrence of different plant species and support different productivity, growth rates, and other factors. While both may contribute some similar ecosystem services, such as contributing to the habitat of a certain species, they are inherently different in their compositions, productivity, and other factors. For any one ecosystem service, such as habitat for one species of

interest, it may be possible to measure the contribution of existing conditions for that one ecosystem service. However, other ecosystem services provided by the site, for example grazing productivity, will be inherently different, so that if the goal is to produce a system that tracks equivalent credit or debit units for a suite of ecosystem services, then use of ecological sites can help assure that equivalent services are capable of being provided. Other ecological classification could serve a similar function. However, other systems are not currently available within the sagebrush biome that consider underlying site potential with the same level of development, mapping, and acceptance by potential users as the NRCS ecological site classification system.

Ecological sites within an MLRA and precipitation zone have been described by NRCS in its ESD process (<http://esis.sc.egov.usda.gov/>). While ESD's have been prepared for many MLRA's within the sagebrush biome, others are still being prepared. NRCS has indicated that completing the ESD's is a high priority, and these should be available in the near future. Where ESD's are lacking,

developers can produce their own descriptions of ecological sites if they have the appropriate knowledge of the ecology of the area. Each ESD provides descriptions of the site, its plant compositions and productivity, soils, and an array of other characteristics. In areas where soils have been mapped, the specific ecological sites occurring on an impact or mitigation area will also be available in a map. In areas where soils have not been mapped, on-site sampling will be needed to determine the specific ecological sites of the impact and mitigation areas based on the soils present in these areas.

Ecological integrity (Karr 2004) refers to an ecosystem's completeness, including the presence of all appropriate components (for example, species), structures (for example, heights of vegetation) and processes such as nutrient cycling or disturbance response. Complete ecosystems possess ecological integrity because they support a biota that is the product of evolutionary and biogeographic processes with minimal changes from human impacts (Karr 2004). Ecological integrity has been a policy objective in several national and bi-national laws and agreements, including the U.S. Water Quality Amendments of 1972 (Clean Water Act), the Great Lakes Water Quality Agreement between the United States and Canada, and the National Wildlife Refuge System Improvement Act of 1997 (Noss 2004).

ESD's generally provide the information needed to develop the description of historical and anthropogenic states. Historical plant communities occurred as a result of disturbance regimes that influenced the composition, structure, and processes of each community. Various historical states typically occurred on any given ecological site. Various additional anthropogenic states that did not occur historically may now occur as a result of human activities over the past 100+ years. Descriptions of the historical states that occurred for each ecological site can be used to provide a baseline reference for comparison. Ecological integrity at a site must be evaluated and quantified relative to an identified reference or baseline to allow an interpretation of its existing and potential future quality for ecosystem services. Historical states define the native ecosystem diversity that occurred within a landscape or MLRA, and provide the best reference for use as baseline conditions for ecological integrity.

Existing Conditions

In addition to the ecological site, the conditions for the existing plant communities occurring on each impact or mitigation site will need to be described through field sampling. The composition of the community as well as other potential measures such as its structure should be sampled and quantified. Measures that are important for

assessing the value of the area for animal species of concern should also be included in this sampling. The existing condition can then be rated as to its ecological integrity in comparison to a reference condition using tools such as a similarity index (NRCS 2003).

Planned development activities (debit units) may be evaluated as to the reduction of ecosystem services associated with the loss of the level of existing ecological integrity to a site, potentially going to zero. Mitigation activities (credit units) may be designed to maintain or increase the ecological integrity of a site and thus its level of ecosystem services. In addition to the ecological site information identified above, information on the existing characteristics of the vegetation at a site is required to complete this evaluation. On each site, the amounts of each plant community and its underlying ecological site provide the basis for the determination of level of ecological integrity. The existing community characteristics are evaluated relative to a reference condition generated from the ESD information to produce a value for the existing ecological integrity, which is then further evaluated relative to landscape context (described below).

Off-site mitigation areas that have equivalent ecological sites could be improved by restoring these areas to conditions that more closely resemble one of the historical states or other desired plant communities based on reference information. The more dissimilar the off-site mitigation area is to the desired plant communities (in other words, high levels of exotic species, low diversity of native species), the greater the potential gain in mitigation value through restoration. Numerous ecosystem services could be increased, including better soil stabilization, improved water quality, increases in grazing land health, and increased quality of fish and wildlife habitat.

Extent of Change

The mitigation metric should measure the change in conditions of plant communities indexed in relation to the historical state or other desired plant community selected as a reference condition. The baseline reference provides a description of the conditions that could be considered optimum in value for a given site, with a possible score of 1 on a 0.1 – 1.0 scale. The existing conditions are then evaluated in relation to this reference condition. If an historical state is desired for use as a baseline reference for calculating credit or debit units, the specific historical state to be used will need to be selected from those that occurred for specific ecological site. Suggestions for making this selection are discussed below in the description of the example.

The metric for a site would evaluate the quality of the existing plant communities at the site in comparison to the reference, and then quantify gains or losses to this quality from impacts or mitigation. For example, a completely degraded site, perhaps dominated by invasive species might score a 0.1 in reference to an identified baseline plant community for a specific ecological site. Impacting this site would not generate a large number of debit units. Mitigation to improve a degraded site to more closely resemble a specified historical state might get partway towards this goal in 3 years, resulting in a 0.6 score for ecological integrity. Each ac improved by this amount would then receive a benefit of the ac times the increase in value, so that in this example, 10 ac increased from 0.1 to 0.6 on a certain ecological site would result in 5 units of improvement (10 ac times 0.5 gain in quality) for each year that the improved condition was maintained. These might offset impacts to other areas of the same ecological site disturbed by energy production or other impacts. In another example, if an existing site had a value of 0.9, and was impacted by an energy development, a large number of debit units might be created. If such a site in an off-site mitigation location was identified and maintained in a high quality condition when it might have decreased to a 0.6 without specific mitigation efforts, then a gain of 0.3 times the area might be identified for each year that the site is maintained in the better condition.

Landscape Context

An important need in determining mitigation metrics is to determine how landscape features influence ecosystem services at a site. The same ecological site and plant community occurring in an impact site or mitigation site may not have the same value or produce the same wildlife benefits through mitigation due to these landscape effects. For example, a development site might be located in mule deer (*Odocoileus hemionus*) winter range. An off-site mitigation site may be identified that has the same ecological site and even existing plant community as the development site, but not be in a mule deer wintering area. The site within the wintering area would be considered of higher conservation value because of this function, even though the site conditions were the same.

Locations of sites important to wildlife species may be identified in two ways. Important habitat for distributions of species of interest may be known and mapped, and this information then used in determining the value of a particular site. Alternatively and more widely used, important sites in a landscape context can be evaluated using habitat modeling. A method of evaluating landscape context is to rely on the expected impacts or benefits for selected wildlife species that are likely to be produced by either impacts or mitigation actions. Models of habitat

quality that include landscape components for selected species of concern could be used to evaluate declines in habitat quality in the development area compared to potential gains in habitat quality for these same species in the mitigation area. Using derivations of habitat suitability index (HSI) models (US Fish and Wildlife Service 1981) that operate at landscape scales, the influence of amounts of specific habitat conditions and their impacts from various developments can be modeled. Further refinements to these models may produce more targeted results. For example, Roloff and Haufler (1997, 2002) developed a habitat-based approach to species viability that provides for the estimation of numbers of home ranges of varying quality for a species in an area to project gains or losses in numbers as a result of development or mitigation activities. Beck and Suring (2008) evaluated the various types of habitat models currently in use in landscape analyses. Through such methods, spatial influences of landscape context can be used as modifiers of the credit or debit units calculated for a specific site.

An important question for evaluating landscape context is which species and how many species of concern should be used as indicators of landscape effects? It would seem desirable to include a number of species within any landscape, but the exact number and which species would need to be determined. If multiple species are used, how should the changes in their habitat qualities be combined to determine the overall landscape context? For example, if off-site mitigation is shown to improve conditions for two sage-brush-associated species but to decrease conditions for another species, how should these be evaluated in the landscape context? Should the net gains and losses of all species be averaged, or are some species more important and therefore receive a higher weighting in the analysis of landscape context? These are policy decisions that need to be determined for the actual application of this mitigation metric system.

The landscape context serves to further adjust the credit or debit units calculated from the site analyses. The site level analysis determines the base level of units. The landscape context then further modifies this level depending on the resulting expected gains or losses to wildlife populations. For example, an impact site might reduce ecological integrity by 50 debit units (for example 100 ac of 0.5 quality reduced to 0.0 quality). A landscape evaluation may then determine that this area supported habitat quality of 0.8 for two species of concern which would drop to a quality of 0.4 with the development. To offset this reduction, a mitigation site that was 100 ac of 0.3 quality would need to be increased to an ecological integrity of 0.8, and be shown to improve habitat quality for the two species by an equivalent increase.

Timing and Duration of Change

Key determinants of the amount of credit or debit units attributed to an activity include expected annual changes, and the total amount of cumulative change (positive or negative) anticipated from the action over some period of time (for example, 20 years), compared to the conditions that would be expected over the same period without the action. Note that under this approach, “preservation” of existing habitat does not gain any credit units unless it eliminates some expected deterioration in condition. “Change” here is intended to refer to changes in the ecosystem services or values provided by the affected areas. As a practical matter, such changes are likely to be inferred from changes in the vegetative characteristics of the affected areas as modified by the landscape level analysis of changes in the use of the area by sagebrush-associated wildlife. These changes need to be evaluated in the context of the extent to which they may impact the flow of services over time relative to the reference condition. In measuring “change” in plant communities, the metric would quantify the amount of change in the plant community over some specified period of time measured in years.

In the sagebrush ecosystem, the impacts of development activities on ecosystem services and species of concern will often be immediate, but they may not always be permanent (for example, well pads and associated roads can be removed after they are no longer in use). Conversely, the benefits of conservation activities on ecosystem services or species of concern will often be neither immediate (for example, restoring sagebrush at dry sites will take many years), nor permanent. Thus, the timing and duration of impacts and conservation activities need to be taken into account when quantifying credits or debits.

Sagebrush Biome Mitigation Metric Example

The above discussion provides a general description of the mitigation metric framework. An example of the application of these concepts is presented here.

Site evaluation and quantification of ecological integrity

The first level of assessment in the mitigation metrics is at the ecological site. The metrics evaluate the ecological integrity of a site (impact or mitigation) in comparison to a reference community. Stated differently, ecological integrity can be linked to the maintenance or restoration of native sagebrush ecosystems because it is recognized that native ecosystems contain the appropriate plant species that are adapted to that specific location and that wildlife species assemblages have developed in response to these adaptations. Definition of a native ecosystem requires reference to what occurred in the area historically, which is typically considered those conditions that occurred prior to

major perturbations brought about following European settlement.

Ecological site descriptions (ESD) (<http://esis.sc.egov.usda.gov/>) use state and transition models to define vegetation states that can occur across sites both historically and under current conditions. Unfortunately, many ESD's only describe one historical state termed the historical climax plant community (HCPC). The HCPC is then described relative to its composition and productivity of native species. However, there are typically multiple historical states that occurred across an ecological site within a landscape depending on the disturbance history of the landscape. For example, figure 1 displays a state and transition model that includes the range of historical states and other anthropogenic states for loamy soils in the Northern Rolling High Plains MLRA (58B) in a 10 to 14” precipitation zone.

Each historical state for an ecological site can be described in terms of the composition, structure, and processes for that historical plant community. For example, a post-fire state dominated by grasses and forbs, and further influenced by light, moderate, or heavy grazing regimes (three different states) might have occurred on a certain ecological site. Over time without fire, sagebrush would be expected to reestablish on the site, resulting in a state with sagebrush intermixed with grasses and forbs, the species of which would be influenced by the level of persistent grazing (figure 1). A reference condition for each of these historical states can be described and used as a basis for comparison to any existing community through use of evaluation tools such as similarity indices. Table 1 provides a description of the compositions of the fire and grazing-influenced historical states included in the state and transition model for loamy sites depicted in figure 1. As a check, a local or regional team of experts may be convened to review the descriptions of historical states for use in mitigation metrics to assure that these descriptions are considered correct and supported by the local natural resource managers.

The proposed metric that addresses ecological integrity compares conditions of current plant communities at a site in relation to a historical state. Because various historical states may have occurred on an ecological site selected for specific mitigation activities, a specific historical state would need to be selected as the baseline reference. The selection of a reference historical state might use either of these approaches:

1. Evaluate which historical states currently have the lowest representation in the surrounding landscape relative to the amounts estimated to have occurred historically, and emphasize restoration of this state.

Such under-represented states provide conditions that are most likely to be limiting to sagebrush-associated species in the landscape because these states are in the smallest amounts relative to their past availability. For example, sagebrush ecosystems in a selected area may have limited amounts of states that contain plant species that decline with grazing pressure, so that the historical states produced under light grazing conditions may be selected as the reference community.

2. Alternatively, a local or regional panel of experts could determine which historical state described for a site is the most important to be emphasized based on the assessment of the most desired vegetation conditions in the landscape either for anticipated ecosystem services or for the importance of that state to the habitat needs of one or more wildlife species of concern.

The existing conditions for the plant community at an impact or mitigation site are measured in the field (for example, vegetation composition and structure). While a standardized sampling methodology is not proposed here, the sampling methods used should be the same for both impact and mitigation sites. Various vegetation metrics could be used to help capture potential changes caused by impacts or mitigation actions. We suggest that a primary measurement would be the relative cover of each plant species. Productivity of each species could also be used, but it is more time consuming to sample, and is complicated by annual fluctuations in productivity based on weather patterns, the different growth rates of woody and herbaceous plants and between grasses and forbs. Additional measures, such as vegetation heights, horizontal cover, or amounts of bare ground could all be potential indicators of existing conditions in comparison to an historical reference.

Baseline or reference conditions of historical states are described from information in the ESDs, with further development based on known or published information on the responses of plant species to different types or levels of disturbance. Development of such information may be reviewed by a local or regional panel of experts with knowledge of plant ecology and range dynamics for the area. Comparisons of existing plant communities can be made to these described historical states through the use of a similarity index (NRCS 2003). A value of an existing plant community may then be calculated for a site that represents the ecological integrity of the ecosystem for that site. From table 1, the reference community that might be selected might be the light grazing, long fire return interval state for loamy sites in the Northern Rolling High Plains, 10

to 14" precipitation zone. Table 2 lists reference standards that might be established for this selected historical state as a reference community. Table 3 lists existing plant community compositions determined from field sampling of 3 specific locations that could serve as either impact or mitigation sites within this MLRA and precipitation zone. Table 4 displays the computed similarity index for ecological integrity of the existing plant communities at these three locations.

The similarity index used the reference community to define an "optimum" composition for desired conditions. The existing community composition is compared to this reference. For each reference standard, the existing community receives points for each percentage of relative cover for the identified native species groups in the reference state, up to the reference standard. Higher amounts than the reference standard of any one group do not add additional points.

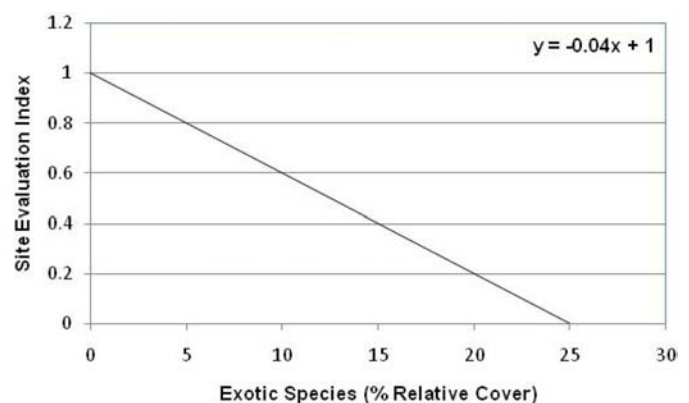


Figure 2—Graph of an example relationship between relative cover of exotic species at a site and the percent reduction in the site similarity score, labeled as the Site Evaluation Index. At 5% relative cover of exotic species, the integrity of a site would be reduced to 0.8% of its value, while at 25% relative cover of exotic species, the integrity of a site would be reduced to 0%. The equation $y = -0.04x + 1$ can be applied to the percentage of exotic species on the site to calculate the exotic species reduction that this line represents.

The ecological integrity of the 3 example sites compared to the reference state is relatively low (table 4) ranging from 0.305 to 0.473. These values are based solely on cover of reference native species found at these sites. An additional modifier of ecosystem integrity might be the level of exotic species occurring on the sites. Figure 2 depicts a possible relationship that could be used to modify a site's ecological integrity based on the relative cover of exotic species sampled for a site. Using this relationship, the adjusted ecological integrity of the 3 sites is listed in table 5.

Table 1—Descriptions of potential historical states occurring on loamy ecological sites in the 10 to 14" precipitation zone of the Northern Rolling High Plains Major Land Resource Area (58B). Letters for each historical state correspond to states shown in figure 1. Scientific names are listed in Appendix A.

Short-Fire Return Interval			
	Light Grazing (A)	Moderate Grazing (B)	Heavy Grazing (C)
Fire Return Interval	<25 years, avg. 5-15 yrs	<25 years, avg. 5-15 yrs	<25 years, avg. 10-20 yrs
Grazing	Intermittent, with rest periods	Variable, but mostly moderate	Mostly season long grazing
Primary Species	Needleandthread, Green needlegrass, Western wheatgrass, Thickspike wheatgrass, Bluebunch wheatgrass, White prairie clover, Prairie coneflower, Dotted blazing star, Winterfat, Indian ricegrass, Cusick's bluegrass, Needleleaf sedge, American vetch, Hawksbeard, Biscuitroot, Evening primrose	Western wheatgrass, Needleandthread, Sandberg bluegrass, Thickspike wheatgrass, Blue grama, Threadleaf sedge, Western yarrow, Indian ricegrass, Bluebunch wheatgrass, Cusick's bluegrass, Needleleaf sedge, Prairie junegrass, Prairie coneflower, White prairie clover, Biscuitroot, Scurfpea, Rosy pussytoes, Milkvetch, Goldenweed, Hawksbeard, Textile onion, Bluebells, Scarlet globemallow, Penstemon, Common pepperweed	Blue grama, Threadleaf sedge, Plains pricklypear, Prairie junegrass, Common yarrow, Rosy pussytoes, Common pepperweed, Western wheatgrass, Thickspike wheatgrass, Sandberg bluegrass, Scurfpea, Milkvetch, Penstemon, Scarlet globemallow, Stemless goldenweed, Textile onion, Bluebells, Hood's phlox
Productivity	1,100 lbs/acre	900 lbs/acre	550 lbs/acre
Structure/Height	5-8	4-6	3-5
Long-Fire Return Interval			
	Light Grazing (D)	Moderate Grazing (E)	Heavy Grazing (F)
Fire Return Interval	>25 yrs	>25 yrs	>25 yrs
Grazing	Intermittent, with rest periods	Variable, but mostly moderate	Mostly season long grazing
Primary Species	Big sagebrush, Needleandthread, Green needlegrass, Western wheatgrass, Thickspike wheatgrass, Bluebunch wheatgrass, White prairie clover, Prairie coneflower, Indian ricegrass, Cusick's bluegrass, Needleleaf sedge, American vetch, Hawksbeard, Biscuitroot, Dotted blazing star, Evening primrose	Big sagebrush, Western wheatgrass, Needleandthread, Sandberg bluegrass, Thickspike wheatgrass, Blue grama, Threadleaf sedge, Western yarrow, Winterfat, Indian ricegrass, Bluebunch wheatgrass, Cusick's bluegrass, Needleleaf sedge, Prairie junegrass, Prairie coneflower, White prairie clover, Biscuitroot, Scurfpea, Rosy pussytoes, Milkvetch, Goldenweed, Hawksbeard, Textile onion, Bluebells, Scarlet globemallow, Penstemon, Common pepperweed	Big sagebrush, Blue grama, Threadleaf sedge, Plains pricklypear, Prairie junegrass, Common yarrow, Rosy pussytoes, Common pepperweed, Western wheatgrass, Thickspike wheatgrass, Sandberg bluegrass, Scurfpea, Milkvetch, Penstemon, Scarlet globemallow, Stemless goldenweed, Textile onion, Bluebells, Hood's phlox
Productivity	925 lbs/acre	750 lbs/acre	475 lbs/acre
Structure/Height	5-8" grasses, sagebrush up to 3'	4-6", sagebrush up to 3'	3-5", sagebrush up to 2.5'

Additional variables could be added to further define important features of a plant community that effectively contribute to the ability of a site to contribute ecosystem services, such as productivity of herbaceous vegetation, horizontal cover for selected wildlife species, or other measures.

These calculations reveal that the 3 sites generally had sufficient sagebrush to meet the desired level of this species identified in the reference state, although site 3 could be improved by slightly increasing the cover of sagebrush. However, all 3 sites had generally low levels of other native species, particularly the "other grass" species that are indicators of a light grazing regime state in this landscape.

Further, the ecological integrity of the sites was dramatically reduced by the level of exotic species.

This example reveals how the quality of a specific location could be improved to gain credit units. By knowing what specific factors kept the site from receiving a higher ecological integrity rating, restoration treatments can be targeted to correct the problem. For example, all 3 sites were very deficient in grasses that decrease in the presence of grazing pressure. Efforts to increase the relative composition of these species on these sites, such as inter-seeding with these species or adjustment of current grazing regimes could be used as treatments to improve this condition. Similarly, all 3 sites lacked sufficient relative

cover of native forbs expected for this plant community, so inter-seeding or other efforts to increase these forbs would be a good treatment to increase the ecological integrity index. Finally, for 2 of the 3 sites, the levels of exotics, especially cheatgrass (*Bromus tectorum*), dramatically decreased the ecological integrity index for the site. Reduction of cheatgrass, possibly through use of selective herbicides, might be an additional mitigation treatment to increase the number of credit units.

Table 2—Reference standards for use in calculating a similarity index of ecological integrity for a loamy ecological site in the 10 to 14" Precipitation Zone of the Rolling High Plains Major Land Resource Area based on use of Historical State D described in table 1 (long-fire return x light grazing historical state) as the reference. Scientific names of species listed in the Appendix A.

Reference Component	Relative Cover	Standard
Big Sagebrush	10-30%	20%
Winterfat	1-10%	5%
Dominant grass (western wheatgrass)	10-40%	25%
Other native grasses (combined; needleandthread, green needlegrass, thickspike wheatgrass, bluebunch wheatgrass, Cusick's bluegrass, Indian ricegrass, and needleleaf sedge)	25-60%	40%
Native forb species (combined; prairie clover, prairie coneflower, American vetch, biscuitroot, hawksbeard, evening primrose, and dotted blazing star)	5-20%	10%

Whether or not the actual weightings and standards used in this example are the preferred values, the example demonstrates how the mitigation metric framework can work at the site level to measure debit and credit units. The example also demonstrates how the determination of these units helps identify the potential management or restoration goals of a mitigation location. It also shows how the framework assures that equivalent ecological sites and resulting plant communities can be compared between impact and mitigation locations.

Landscape Context

A goal of the mitigation metrics framework is to develop a process that promotes the mitigation of ecosystem services in the sagebrush biome, including maintaining or improving conditions for sagebrush-associated species. To address the needs of such species, analysis of impact or mitigation assessments are needed at the landscape level. This is

proposed to be accomplished through the use of habitat models for selected sagebrush-associated species.

Table 3—Relative cover of plant species for three example locations sampled within the 10 to 14" precipitation zone of the Northern Rolling High Plains Major Land Resource Area (58B). Scientific names of species listed in the Appendix.

Plant Species	Site		
	1	2	3
Relative cover (percent)			
American vetch	0.0	0.4	0.0
Cheatgrass ^a	14.8	26.5	3.9
Big sagebrush	24.4	25.6	13.2
Blue grama	4.5	1.2	8.2
Sandberg bluegrass	25.8	6.3	1.9
Canadian horsetweed	0.0	0.2	0.0
Common dandelion ^a	0.0	0.4	0.0
Common	0.0	0.2	0.1
Desert madwort ^a	0.2	0.2	0.0
False pennyroyal	1.2	0.9	0.0
Knotweed	0.2	0.4	0.0
Narrowleaf four	0.0	0.2	0.0
Needle and thread	1.4	5.9	9.1
Needleleaf sedge	4.1	0.5	11.5
Plains pricklypear	1.2	0.9	1.4
Prairie junegrass	5.7	3.0	0.5
Prairie sagewort	0.2	0.0	0.5
Purple threeawn	0.5	1.2	0.5
Rosy pussytoes	0.0	0.2	0.0
Scarlet globemallow	0.7	0.0	0.5
Sixweeks fescue	3.6	3.0	10.1
Textile onion	0.0	0.2	0.0
Threadleaf sedge	0.0	3.3	0.0
Western wheatgrass	8.1	16.2	38.5
Winterfat	1.0	2.6	0.0
Wooly plantain	2.1	0.9	0.1
Yellow salsify ^a	0.2	0.0	0.0
Total Exotic Species	15.2	27.1	3.9

^aExotic species

Habitat models that can operate effectively and be endorsed by experts on sagebrush-associated species for use at the landscape level are not widely available at the present. While numerous habitat models exist for conservation planning at the landscape scale (Beck and Suring 2008), obtaining agreement for their use relating to energy developments in the sagebrush biome has proven to be a more complicated process. Here, we will not present the pros or cons of any specific models, but will discuss how models can serve to evaluate the landscape context for the mitigation metric framework. Models for use in the

mitigation metric framework should be acceptable to the mitigation decision-making authority, identify the specific habitat information for evaluating the contribution of impact and mitigation sites, and provide a basis for delineation of the landscape to be used in the evaluation.

Use of specific models will require maps of vegetation and other landscape attributes that provide the required input data of sufficient accuracy and detail to run the selected models. Availability of such maps and associated habitat variables are one of the most limiting features for running habitat models at landscape scales. Use of even the most sophisticated habitat models will be unlikely to produce accurate results if the quality of the underlying vegetation map including an assessment of its ability to provide the specific information required by the model is not addressed.

Various habitat models for sage-brush associated species exist. Considerable attention has recently been directed at developing habitat or impact models for greater sage-grouse because of the concerns about the status of this species. Models for other sage-brush associated species exist, but most lack the level of detail, testing, and data generation that has been directed at sage-grouse models. While selection of the specific species to be included in the landscape assessment will be determined by the mitigation-decision authority, we suggest that 5 to 10 sagebrush-associated species be used in evaluating the landscape context. These most often may include sage-grouse, pronghorn antelope (*Antilocapra americana*), and mule deer because of the large home range requirements of these species and the public interest and concern about their status. Additional sagebrush-associated species that include a range of required sagebrush conditions or historical states should also be considered. An example of some of the habitat models available for use in the sagebrush biome is included in Rowland and others (2006).

The species and the habitat models selected for use in the landscape evaluation for mitigation metrics will determine the specific habitat variables that need to be considered. Those habitat variables that relate to vegetation characteristics should be identified, and the required information included in vegetation sampling conducted at potential development or mitigation sites. This will insure that the actual contributions of these sites to the habitat requirements of selected sagebrush-associated species is known and documented.

The area to be included in the landscape assessment will depend on the setting and the needs of the species selected for the landscape context. Some resident species have small home ranges, and their complete habitat needs can be

assessed with a relatively small landscape context. Other resident species, including some populations of sage-grouse, mule deer, and pronghorn antelope, will make seasonal movements and the assessment should include their entire annual home ranges. This will mean that the landscape for some evaluations will be quite large.

Implementation of a Mitigation Metrics Framework

The mitigation metrics framework provides an index of ecosystem services provided per ac that can be used to determine the number of credit and debit units. The credits or debits are based on the ecological site classification system of NRCS. This classification system describes ESDs that occur within Precipitation Zones which then overlap with MLRA's. Credit and debit units are therefore defined for specific Ecological Sites within a Precipitation Zone and MLRA. This combination of areas then defines the minimum operational area for the mitigation metric system. This framework assures that credit and debit units are equal at the site level. The additional assessment at the landscape scale assures that credits and debits are also equal for species influenced by surrounding landscape factors.

There may be opportunities to provide mitigation benefits that do not correspond to this strict compliance of similar ecosystem services. For example, there may be advantages to addressing impacts occurring in one MLRA through off-site mitigation in an adjacent or nearby MLRA. In this case, the assumptions concerning the equality of ecosystem services may not hold. This should not preclude the consideration of such mitigation opportunities, but decisions to allow such trade-offs of credits and debit units should be made with the clear recognition that equivalency may not be achieved. Similarly, even within one MLRA, it may be desirable to mitigate impacts to one ecological site on a different ecological site. For example, impacts to a saline upland site might be more efficiently off-set and landscape context considerations better addressed by conducting mitigation in a loamy site that has higher productivity and greater diversity of plant species. However, in both of these examples, careful consideration should be given to the cumulative impacts that might be occurring to one particular ecological site, and sufficient consideration given to maintaining enough area of high ecological integrity for that site within the MLRA. If tradeoffs in types of sites are considered acceptable to a mitigation-decision authority, a general suggestion would be that sites with higher productivity or diversity of species should not be allowed to be mitigated through use of sites with a lower productivity or diversity of species.

Table 4—Example calculation of a similarity index for ecological integrity of existing plant communities sampled on loamy sites in the 10 to 14” precipitation zone of the Northern Rolling High Plains Major Land Resource Area (58B). Scientific names of species are listed in the Appendix.

Species Group	Reference Standard from Table 2	Site 1		Site 2		Site 3	
		Cover Data (%)	Index Score	Cover Data (%)	Index Score	Cover Data (%)	Index Score
Big sagebrush	20%	24.4%	0.200	25.6%	0.200	13.2%	0.132
Winterfat	5%	1.0%	0.010	2.6%	0.026	0%	0.000
Western wheatgrass	25%	8.1%	0.081	16.2%	0.162	38.5%	0.250
Other grasses for this historical state	40%	1.4%	0.014	5.9%	0.059	9.1%	0.091
Forbs for this historical state	10%	0%	0.000	0.4%	0.004	0%	0.000
Ecological Integrity Score			0.305		0.451		0.473

Table 5—Exotic species modifier for ecological integrity scores for example plant communities sampled in the 10 to 14” precipitation zone of the Northern Rolling High Plains Major Land Resource Area (58B). Similarity scores are from table 3. Modifiers for levels of exotic species are from figure 2.

	Site 1	Site 2	Site 3
Site Similarity Index Score	0.305	0.451	0.473
Level of Exotic Species (%) from table 3	15.2	27.1	3.9
Integrity Score Modifier from figure 2	0.392	0.000	0.844
Adjusted Ecological Integrity Score	0.120	0.000	0.399

An additional consideration is how area and quality may interact. Impacts to a very high quality site in terms of its existing ecological integrity theoretically could be off-set by increasing the ecological integrity of additional ac of sites with very low ecological integrity to a moderate level of integrity. While mathematically this may work, the biological response may be very different. The implications of such tradeoffs should be considered and may require additional standards so that cumulative impacts to high quality sites do not result in substantial reductions of these conditions.

As mentioned in the landscape context section, which species to consider and the number of species to consider in evaluating landscape influences is a determination that must be made. Further, the combined response of multiple species in the landscape assessment needs to be determined.

MONITORING

The scope of monitoring instituted in association with mitigation metrics will be up to the mitigation-decision authority determining appropriate mitigation standards.

Monitoring is strongly recommended to track expected credit units following mitigation treatments. Some mitigation-decision authorities may require that the actual changes in plant communities be produced and documented through sampling prior to awarding credit units. Other mitigation authorities may acknowledge credits once treatments have been applied, while others may acknowledge credits with only a commitment to conduct restoration treatments. The mitigation metric framework provides a strong basis for developing a specific restoration treatment and monitoring program. The level and kind of monitoring may vary by individual project.

An effective monitoring program should be sufficiently flexible to allow modifications as new findings are produced, but require sufficient standardization to be credible and reproducible. We encourage use of an adaptive management design (Holling 1978), so that specific mitigation metrics can be continually evaluated and adjusted as needed to make them as accurate and effective as possible. Standard survey procedures or other previously established monitoring protocols based on sound science should be used in this process. Though the specific monitoring for each project may differ, some factors that may be important to monitor include vegetative responses to management, the presence of invasive species, and presence or abundance of species of concern.

CONCLUSIONS

The mitigation metrics framework described here provides a strong foundation for determining and measuring equivalency of mitigation credit and debit units at the site level. It also outlines a mechanism for incorporating a standardized method of evaluating landscape context and influences. The application at the site level is fairly straightforward, and the information on ESD's and soils maps are available in many areas. Providing this information in remaining areas is a priority for NRCS. Even where the

information is not available, the mitigation metrics can be efficiently developed and applied by knowledgeable experts.

Landscape context remains a larger challenge. Identifying which species to include, which species models are acceptable, and developing the required data and maps to accurately run the models at the landscape scale can be difficult. This challenge is not unique to this mitigation metric framework. Rather, it is an over-arching concern for impact assessment and mitigation assessment within the sagebrush and other biomes.

ACKNOWLEDGEMENTS

The Cooperative Sagebrush Initiative encouraged the development of this mitigation metrics framework. Input to this development was provided by numerous participants at a workshop on creating a credit trading system for mitigation in the sagebrush biome. Michael Bean and Jay Hestbeck provided input to an earlier document on development of the mitigation metrics. Amy Ganguli developed the reference community descriptions for work conducted for the Thunder Basin Grasslands Prairie Ecosystem Association.

REFERENCES

- Beck, J.L., Suring, L.H. 2009. Wildlife habitat-relationships models: description and evaluation of existing frameworks. Pages 251-285 in Millsaugh, J.J., Thompson, F.R., III, editors. Models for planning wildlife conservation in large landscapes. Elsevier Science, New York.
- Friedel, M.H. 1991. Range condition assessment and the concept of thresholds: a viewpoint. *Journal of Range Management* 44:422-426.
- Holling, C.S., editor. 1978. Adaptive environmental assessment and management. John Wiley, New York, New York, USA.
- Karr, J.R. 2004. Beyond definitions: maintaining biological integrity, diversity, and environmental health in National Wildlife Refuges. *Natural Resources Journal* 44:1067-1092.
- Laycock, W.A. 1991. Stable states and thresholds of range condition on North American rangelands: a viewpoint. *Journal of Range Management* 44:427-433.
- Noss, R.F. 2004. Some suggestions for keeping National Wildlife Refuges healthy and whole. *Natural Resources Journal* 44:1093-1111.
- Roloff, G.J., Haufler, J.B. 2002. Modeling habitat-based viability from organism to population. Pages 673-686 in: Scott, M.J., Heglund, P.J., Morrison, M.L., Haufler, J.B., Raphael, M.G., Wall, W.A., Samson, F.B. editors. Predicting species occurrence. Island Press, Washington, D.C.
- Roloff, G.J., Haufler, J.B. 1997. Establishing population viability planning objectives based on habitat potentials. *Wildlife Society Bulletin* 25: 895-904.
- Rowland, M.M., Leu, M., Hanser, S., Finn, S.P., Aldridge, C.A., Knick, S.T., Suring, L.H., Boyd, J.M., Wisdom, M.J., Meinke C.J. 2006. Assessment of threats to sagebrush habitats and associated species of concern in the Wyoming Basins. Version 2.0, March 2006, unpublished report on file at USGS Biological Resources Discipline, Snake River Field Station, 970 Lusk St., Boise, ID 83706.
- USDA Natural Resources Conservation Service (NRCS). 2006. Land resource regions and major land resource areas of the United States, the Caribbean, and the Pacific Basin. USDA Handbook 296.
- USDA Natural Resources Conservation Service (NRCS). 2003. National range and pasture handbook. USDA Natural Resources Conservation Service 190-VI-NRPH.
- U.S. Fish and Wildlife Service. 1981. Standards for the development of habitat suitability index models for use in the Habitat Evaluation Procedures. U.S. Fish and Wildlife Service, Division of Ecological Services, ESM 103, Washington, D.C.

Appendix A—Scientific names of plant species listed in Tables. Nomenclature after *The PLANTS Database* (<http://plants.usda.gov>; accessed 12/2006). Origin refers to N= native, I= introduced

Family Symbol	Scientific Name	Common Name	Origin
Apiaceae			
LOSP	<i>Lomatium</i> spp.	biscuitroot	N
Asteraceae			
ACMI2	<i>Achillea millefolium</i>	common yarrow	N
ANRO2	<i>Antennaria rosea</i>	rosy pussytoes	N
ARFR4	<i>Artemisia frigida</i>	prairie sagewort	N
ARTR2	<i>Artemisia tridentata</i>	big sagebrush	N
COCA5	<i>Conyza canadensis</i>	Canadian horseweed	N
CRSP	<i>Crepis</i> spp.	hawkbeard	I
RASP	<i>Ratibida</i> spp.	Prairie coneflower	N
STAC	<i>Stenotus acaulis</i>	Stemless goldenweed	I
Boraginaceae			
MESP	<i>Mertensia</i> spp	bluebells	N
Brassicaceae			
ALDE	<i>Alyssum desertorum</i>	desert madwort	I
LEDE	<i>Lepidium densiflorum</i>	common pepperweed	N
Cataceae			
OPPO	<i>Opuntia polyacantha</i>	plains pricklypear	N
Chenopodiaceae			
KRLA2	<i>Krascheninnikovia lanata</i>	winterfat	N

Appendix A (Continued) —Scientific names of plant species.			
Family Symbol	Scientific Name	Common Name	Origin
Cyperaceae			
CADU6	<i>Carex duriuscula</i>	needleleaf sedge	N
CAFI	<i>Carex filifolia</i>	threadleaf sedge	N
Fabaceae			
ASSP	<i>Astragalus</i> spp.	milkvetch	N
DACA7	<i>Dalea candida</i>	white prairie clover	N
PSSP	<i>Psoralea</i> spp.	scurfpea	N
VIAM	<i>Vicia americana</i>	American vetch	N
Lamiaceae			
HEHI	<i>Hedeoma hispida</i>	false pennyroyal	N
Liliaceae			
ALTE	<i>Allium textile</i>	textile onion	N
Malvaceae			
SPCO	<i>Sphaeralcea coccinea</i>	scarlet globemallow	N
Nyctaginaceae			
MILI3	<i>Mirabilis linearis</i>	narrowleaf four o'clock	N
Onagraceae			
OESP	<i>Oenothera</i> spp.	evening-primrose	N
Plantaginaceae			
PLPA2	<i>Plantago patagonica</i>	woolly plantain	N
Poaceae			
ACHY	<i>Achnatherum hymenoides</i>	Indian ricegrass	N
ANBR	<i>Annual brome</i>	cheatgrasses	I
ARPU9	<i>Aristida purpurea</i>	purple threeawn	N
BOGR2	<i>Bouteloua gracilis</i>	blue grama	N
ELLA	<i>Elymus lanceolatus</i>	thickspike wheatgrass	N
NAVI4	<i>Nassella viridula</i>	green needlegrass	N
PASM	<i>Pascopyrum smithii</i>	western wheatgrass	N
POCU3	<i>Poa cusickii</i>	Cusick's bluegrass	N
POSE	<i>Poa secunda</i>	Sandberg bluegrass	N
PSSP6	<i>Pseudoroegneria spicata</i>	bluebunch wheatgrass	N
VUOC	<i>Vulpia octoflora</i>	sixweeks fescue	N
Polemoniaceae			
PHHO	<i>Phlox hoodii</i>	Hood's phlox	N
Polygonaceae			
POSP	<i>Polygonum</i> spp.	knotweed	N
Santalaceae			
PESP	<i>Penstemon</i> spp.	penstemon	N

Copies of Natural Resources and Environmental Issues are available for the S.J. and Jessie E. Quinney Natural Resources Research Library, College of Natural Resources, Utah State University, Logan UT 84322-5260. Unless otherwise noted, single issues are \$20.00 in North America and \$25.00 elsewhere. Please make payments with a check, money order, or purchase order, payable in U.S. dollars to NREI. Online order forms can be found at <http://cnr.usu.edu/quinney/htm/publications/nrei>.

- Vol. A Wilderness Areas. Edited by Allen Rasmussen. 1990. \$8.00
- Vol. I Riparian Resources. Edited by G. Allen Rasmussen and James P. Dobrowolski. 1994. \$12.00
- Vol. II Mapping Tomorrow's Resources. Edited by Allen Falconer. 1993
- Vol. III Conflicts in Natural Resources Management. Edited by Joanna Endter-Wada and Robert Lilieholm. 1995
- Vol. IV Biodiversity on Rangelands. Edited by Neil West. 1995
- Vol. V Ecosystem Management of Natural Resources in the Intermountain West. Edited by Frederic H. Wagner. 1995
- Vol VI Unavailable
- Vol. VII Proceedings of the 2nd Biennial Conference of University Education in Natural Resources. Compiled by Carla G. Heister. 1998
- Vol. VIII SwarmFest 2000: Proceedings of the 4th Annual Swarm User Group Meeting. Edited by William C. Pitt. 2001
- Vol. IX Proceedings of the 4th Biennial Conference of University Education in Natural Resources. Compiled by Gary B. Blank. 2002
- Vol. X Wolves in Utah: An Analysis of Potential Impacts and Recommendations for Management. Prepared by T.A. Switalski, T. Simmons, S.L. Duncan, A.S. Chaves, and R.H. Schmidt. 2002. \$8.00
- Vol. XI Galaxy II Conference: Exceeding Expectations Through Teamwork. Edited by Robert Heister and Compiled by Patsy Palacios. 2004
- Vol. XII Proceedings of the 5th Biennial Conference of University Education in Natural Resources. Compiled by Tom Kold. 2004
- Vol. XIII Water in the West: Use-Abuse-Scarcity (DVD). Sponsored by Chris Luecke, David Tarboton, and Patsy Palacios. 2006
- Vol. XIV Bear Lake Basin: History, Geology, Biology, and People. By Patsy Palacios, Chris Luecke, and Justin Robinson. 2007
- Vol. XV Saline Lakes Around the World: Unique Systems with Unique Values. Edited by Aharon Oren, David Naftz, Patsy Palacios, and Wayne A. Wurtsbaugh. 2009.

NREI is an open access journal which means that all content is freely available without charge to the user or his/her institution. Users are allowed to read, download, copy, distribute, print, search, or link to the full texts of the articles in this journal without asking prior permission from the publisher or the author. This is in accordance with the BOAI definition of open access. Visit us at <http://digitalcommons.usu.edu/nrei/>.

