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THREATS TO SHRUBLAND ECOSYSTEM INTEGRITY

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Abstract

Abstract: The 29 papers in this proceedings are divided into the main organized sessions of the 16th Wildland Shrub Symposium, including the plenary session to introduce the theme of threats to shrubland ecosystem integrity, impacts of energy development and reclamation on ecosystem function, invasive plant ecology. wildlife habitats: impacts and restoration opportunities, historical perspectives in shrublands, ecosystem threats due to fire in the Mojave Desert, and modeling and monitoring of shrubland ecosystems. An overarching goal of the symposium was to make linkages between research and management.

Keywords: wildland shrubs, wildlife habitat, disturbance regimes, restoration, monitoring, energy development impacts

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Sagebrush taken by Thomas Monaco



16th Wildland Shrub Symposium
Threats to Shrubland Ecosystem Integrity
2010 May 18-20
Logan, UT

Compilers:

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Series Forward

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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.

Fifteenth: 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. http://www.fs.fed.us/rm/pubs_other/rmrs_2011_wambolt_c001.pdf.

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Is Climate Change Mitigation the Best Use of Desert Shrublands?

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ABSTRACT

In a world where the metrics of the carbon economy have become a major issue, it may come as a surprise that intact cold desert shrublands can sequester significant amounts of carbon, both as biomass and in the form of SOC (soil organic carbon). Xerophytic shrubs invest heavily in belowground biomass, placing fixed carbon in an environment where it turns over only very slowly. In order for humans to gain this important ecosystem service, desert shrublands must be kept intact and prevented from frequent burning. The biggest threat to shrubland integrity is the invasion of exotic annual grasses that increase fire frequency to the point that most shrubs can no longer persist. Not only do annual grasslands sequester very little carbon, they also increase the turnover rate of existing SOC. From the point of view of carbon sequestration, restoring the many millions of hectares of annual grass dysclimax in the Interior West to functioning shrubland ecosystems should have high priority. The elimination of perennial understory vegetation and cryptobiotic crusts is a nearly inevitable consequence of livestock grazing in deserts. This opens these systems to annual grass invasion, subsequent burning, and loss of a major carbon sink, a heavy price to pay for the minimal economic gains derived from direct use of these intrinsically unproductive lands for livestock production. On a more immediate scale, the conversion of stable desert shrublands to annual grasslands that burn frequently has also created major issues with windblown dust. Good evidence exists to show that deposition of this dust on mountain snowpack can have the effect of reducing water yield by causing premature melting. Water is clearly the most limiting resource for agriculture in our region, and protecting mountain watersheds from dust deposition should become another important priority. As climate disruption in all its forms becomes a major threat to production agriculture, it is imperative that serious steps be taken to minimize this threat, including restoration of degraded shrubland ecosystems, and prevention of degradation of shrublands that are still intact. Here the argument is made that the best use of cold desert shrublands is mitigation of both short term and long term climate disruption.

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INTRODUCTION

Deserts and semideserts occupy approximately 22 percent of the earth's land surface (Janzen 2004), yet because of their low productivity, they are generally assumed to be relatively minor players in the global carbon cycle. Schemes to mitigate global climate change have rarely included the idea that improving carbon sequestration in deserts could make a significant contribution at a global scale. Many ideas for increasing carbon sequestration, such as tree plantations in marginally suitable environments, involve tradeoffs with other resource values such as water use and quality (Jackson and others 2005). In contrast, improving carbon sequestration in deserts by restoring degraded shrublands to a more functional state would address a broad suite of resource values, including improved air and water quality, wildland fire abatement, enhanced wildlife habitat, biodiversity conservation, and aesthetic and recreational values.

The question addressed here is whether such restoration on a broad scale in the interior West could also make a significant contribution to climate change mitigation. The premise is that restoration of degraded cold desert shrublands could result in sequestration of significant amounts of carbon, and could also reduce the negative climatic effects of excessive windblown dust. The consumptive uses of these ecosystems, which could potentially interfere with management for carbon sequestration, could be said to be relatively unimportant economically, at least in the Interior West. If the carbon credit market that is currently taking shape internationally becomes fully functional, well-managed cold deserts may be able to provide more revenue as carbon sinks than as grazing lands. In addition, management for carbon sequestration can also be viewed as management for maximum return in terms of many other ecosystem services and amenity resources.

Carbon Storage In Deserts

Examination of carbon (C) storage patterns in major biomes on a global scale reveals that deserts (including semideserts) are responsible for the storage of a substantial proportion of the terrestrial C pool (table 1). Stored carbon may be present as standing biomass or as soil organic carbon (SOC), with SOC generally considered to be the more stable and persistent form. It dominates the terrestrial carbon pool at about 80 percent of total stored C (Janzen 2004). The relative contribution of C as standing biomass versus SOC in deserts is even more strongly biased, with over 95 percent of the stored C as SOC. Standing biomass C in deserts is estimated to account for only 1.7 percent of global total, whereas desert SOC is estimated to account for 9.5 percent. Overall, deserts account for about 8 percent of terrestrial C stocks (Janzen 2004). This indicates that deserts are generally about a third as effective as the average biome at storing C on a per area basis. Given the intrinsically unproductive nature of deserts, these figures at first seem surprising. It is hard to see how systems that support such low standing biomass can generate so much SOC. But the same factor that generally limits biomass production in deserts, namely lack of water during much of the year, particularly when temperatures are warm, also limits the rate of microbial respiration in soil, leading to accumulation and persistence of SOC (Jobbagy and Jackson 2000).

The vertical distribution of C in deserts also helps explain how they can be effective carbon sinks (figure 1). When compared with other temperate region biomes, standing biomass, particularly in cold deserts, is dominated by the belowground portion, with root: shoot ratios averaging between four and five (Jackson and others 1996; figure 2). The maximum rooting depth is deeper for cold deserts than for any other biome examined (Canadell and others 1996), and less than 55 percent of root biomass is found in the upper 30 cm of soil (Jackson and others 1996).

This contrasts with perennial grasslands, which have similar standing biomass and relatively high root: shoot ratios, but with >80 percent of the root biomass in the surface 30 cm. This pattern of deep and extensive rooting in cold deserts is probably related to the need to capture winter precipitation stored at depth during the ensuing growing season, which is usually quite dry. The pattern is not seen in warm deserts, where summer monsoonal moisture patterns dominate and root: shoot ratios average less than one (Jackson and others 1996). In deserts, and in shrublands in general, SOC and standing belowground biomass follow similar distribution patterns, that is, with more SOC in deeper soil layers relative to the surface layer than is found in either grassland or forest vegetation (Jobbagy and Jackson 2000). The estimated proportion of total SOC found from 1-3 m in depth is higher for deserts (0.86) than for any other temperate ecosystem.

Table 1. Estimated terrestrial global carbon stocks by biome (Janzen 2004) and estimated mean carbon stock per unit area for each biome.

Biome	Area (10 ⁹ ha)	Global Carbon Stocks (Pg)			Carbon stock/area
		Plants	Soil	Total	
Temperate Forests	1.04	59	100	159	152.9
Boreal Forests	1.37	88	471	559	111.6
Temperate Grasslands/Shrublands	1.25	9	295	304	89.3
Deserts and Semideserts ¹	3.04	8	191	199	58.2
Tundra	0.95	6	121	127	17.9
Croplands	1.60	3	128	131	81.9
 Tropical Forests	 1.76	 212	 216	 428	 243.2
Tropical Savannas/Grasslands	2.25	66	264	330	108.1
Wetlands	0.35	15	225	240	68.6
 Total (not including ice cover)	 13.61	 466	 2011	 2477	 182
 % of total in deserts/semideserts	 22.3%	 1.7%	 9.5%	 8.0%	

¹Area and carbon stock per area estimates in Janzen (2004) for the desert/semidesert biome have been adjusted by removal of areas of ice cover.

In general, SOC has a deeper distribution in soil than roots, and this is especially true in ecosystems with lower precipitation. The most likely explanation for this is that SOC turnover at depth is very slow. Dominance of more slowly degrading forms of carbon, lower nutrient concentrations, and more resistant root tissues at depth contribute to SOC persistence (Jobbagy and Jackson 2000).

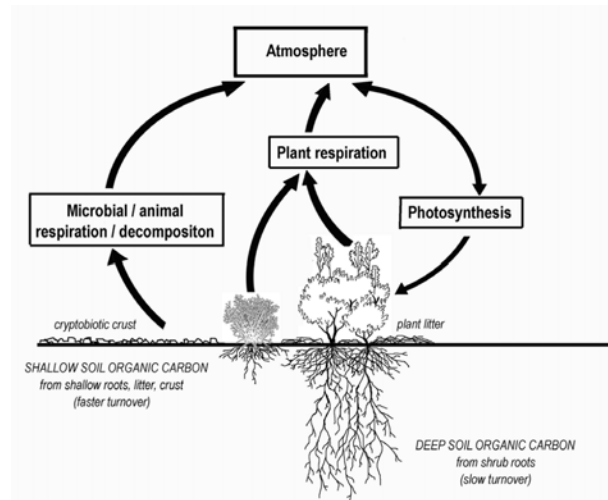


Figure 1. The carbon cycle in a cold desert ecosystem, showing fluxes to the atmosphere (plant respiration and animal/microbial respiration/decomposition), uptake from the atmosphere by plants (primarily shrubs and grasses; photosynthesis), standing plant biomass, and shallow and deep soil organic carbon (SOC). If C uptake exceeds C flux to the atmosphere, C sequestration to a net carbon sink takes place, whereas if flux to the atmosphere exceeds uptake, the system functions as a net carbon source. Deep SOC (soil organic carbon), the most stable form of stored C, dominates C storage in deserts and semideserts.

The ability of cold desert soils to retain SOC could be reduced by the effects of ongoing climate change. Aanderud and others (2010) showed in an 11-year rain manipulation study that near-surface (0-30 cm) SOC stocks in a sagebrush steppe (*Artemisia tridentata*) community were significantly reduced when precipitation was shifted from a winter pattern to a spring-summer pattern. They credited this loss to increased microbial activity in wet surface soil at warm temperatures. Shifts from winter to spring-summer rainfall patterns are predicted for many parts of the Interior West as climate continues to warm (Zhang and others 2007). Rainfall timing impacts on deep SOC would be expected to be lower, however, because deep SOC is more buffered from seasonal

temperature changes. This would tend to mitigate the effects of increased warm-season precipitation on soil C storage.

Carbon cycling on US rangelands has been the subject of several recent studies and reviews (e.g., Bird and others 2002, Hunt and others 2004, Schuman and others 2002, Svejcar and others 2008, Follett and Reed 2010, Brown and others 2010). Synthesis of information on carbon storage on rangelands is complicated by the fact that many different vegetation types occurring under many different climatic regimes fall under the rubric of rangelands. Hunt and others (2004), working in Wyoming, found that mixed grass prairie vegetation was carbon-neutral, whereas sagebrush steppe vegetation was acting as a carbon sink. Schuman and others (2002) focused on the potential to increase carbon sequestration in rangelands through improved management, particularly grazing management. Their emphasis was primarily on grasslands. Svejcar and others (2008) report the results of a very interesting 6-year study on net ecosystem C exchange at eight rangeland sites across a range of habitats. They found that both sagebrush steppe sites and three of four perennial grassland sites generally acted as C sinks during the course of the study, whereas the two warm desert sites acted as C sources. Whether a site acted as a source or a sink varied across years and was closely tied to precipitation patterns. Drought years limited productivity and tended to make even the most productive sites temporary C sources.

Because cold deserts store much of their carbon belowground, and because the carbon is stored in deeper soil layers, these deserts are likely to store more carbon per unit area than warm deserts with monsoonal moisture regimes. In addition, the desert shrublands of the interior West might be more appropriately classified as semideserts, as they generally have much higher standing biomass than the true deserts, for example, the Sahara Desert of North Africa, which is virtually plantless over large areas except in drainageways (wadis). This combination of high belowground allocation and relatively high biomass production appears to make cold deserts exceptionally good candidates for management for carbon sequestration.

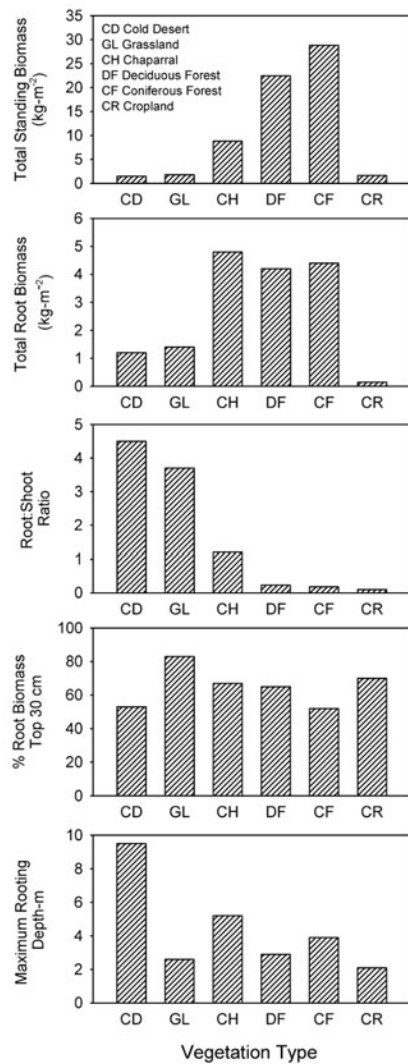


Figure 2. Quantity and distribution of biomass carbon in cold desert biomes contrasted with other temperate zone biomes (grassland, chaparral, deciduous forest, coniferous forest, and cropland): A) total standing biomass, B) total root biomass, C) root:shoot ratio, D) % root biomass in the top 30 cm, and E) maximum rooting depth (adapted from Jackson and others 1996).

Shrubland Degradation and Carbon Storage

Historically, intact desert ecosystems were most likely in a steady state relationship with regard to carbon budgets, acting in the long term neither as sources nor sinks. But two sets of factors have been operating to disturb this steady state, and these factors generally operate in opposing directions. First, woody 'encroachment' of former desert and other temperate grasslands is often thought to have shifted

the carbon balance in these ecosystems to make them net carbon sinks. Whether conversion from perennial grassland to woody vegetation results in a net increase in C sequestration is the subject of considerable debate, however. Jackson and others (2002) found that whether woody encroachment of perennial grasslands resulted in an increase or decrease in SOC depended on precipitation. There was substantial loss of SOC with woody encroachment in more mesic environments, a loss sufficient to more than counterbalance the increase in standing biomass C resulting from the conversion to dominance by woody species. At the dry end of the spectrum, on the other hand, conversion from perennial desert grassland to shrubland resulted in increases in both standing biomass C and SOC. Most land managers regard woody encroachment as a form of degradation, but its causes are complex and in many cases not completely understood. Climate change may itself be driving woody encroachment in some ecosystems, for example, in the northern Chihuahuan Desert, where creosote bush (*Larrea tridentata*) and tarbush (*Flourensia cernua*) are actively invading desert grasslands (Van Auken 2000). Changes in historic fire regimes, poor grazing management, and other factors may contribute to woody encroachment in other semiarid ecosystems, for example, the invasion of juniper (*Juniperus* spp.) species into sagebrush steppe in the Interior West.

The second process that has had a major impact on carbon storage in the deserts of western North America is the displacement of desert shrubs by invasive annual grasses through increased frequency of fire following destruction of the perennial herbaceous understory through improper grazing management. This phenomenon has not received the attention of carbon brokers that has been given to woody encroachment, but it potentially has more impact on carbon budgets, as it is very likely in the process of converting large portions of the Great Basin and surrounding areas into carbon sources. This possibility was apparently first noted by Bradley and coworkers (Bradley and Mustard 2005, Bradley and others 2006). Using sophisticated remote sensing technologies, these authors conservatively estimated that the area of former salt desert and shrub steppe vegetation in the Great Basin alone that has been converted through repeated burning to cheatgrass monocultures as of 2006 was on the order of 20,000 km². In addition, cheatgrass is not the only invasive annual grass that is having major impacts in western

North America. Medusahead wildrye (*Taeniatherum caput-medusae*) and North Africa grass (*Ventenata dubia*) are major invaders in the Interior Northwest, while red brome (*Bromus rubens*) has become a driver of frequent large-scale fires in the Mojave Desert. Many of these fires are occurring in fire-intolerant shrub communities, for example, blackbrush (*Coleogyne ramosissima*) shrublands, that had very low pre-invasion probabilities of burning (Brooks and others 2004).

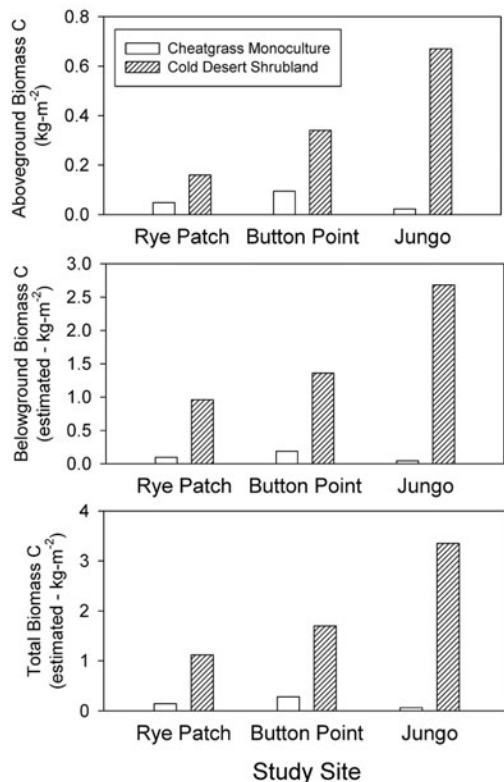


Figure 3. Standing biomass carbon in intact cold desert shrubland communities versus adjacent areas that have been converted to cheatgrass (*Bromus tectorum*) monocultures at Rye Patch NV (salt desert shrubland), Button Point NV (sagebrush steppe), and Jungo NV (sagebrush steppe). Aboveground biomass data from Bradley and others (2006); belowground and total biomass estimated from independent root:shoot ratio data.

Bradley and others (2006) also carried out an on-the-ground assessment of carbon stocks in cold desert shrublands versus cheatgrass monocultures. They measured above-ground carbon stocks and SOC in the near-surface soil horizon in burned and unburned salt desert shrubland (one site) and Wyoming big sagebrush steppe (two sites). They demonstrated a

three- to thirty-fold decrease in standing aboveground carbon stocks as a consequence of type conversion to cheatgrass (figures 3 & 4).

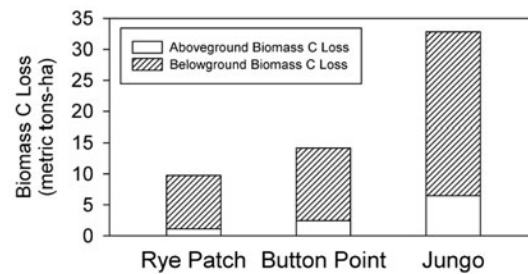


Figure 4. Estimated loss of biomass carbon resulting from conversion from cold desert shrubland to cheatgrass (*Bromus tectorum*) monoculture at three Nevada sites (adapted from Bradley and others 2006; see text for details).

While the study of Bradley and others (2006) did not include any assessment or estimate of root biomass C, root:shoot ratio information for the dominant species obtained from other studies can provide at least a rough estimate of root biomass C in these communities. The estimate of two used here for the root:shoot ratio for cheatgrass is undoubtedly high; in greenhouse and field studies, root:shoot ratios greater than one for this species are rarely encountered, but a conservative estimate was chosen for purposes of avoiding exaggeration of differences (Meyer unpublished data). The estimate of six for the root:shoot ratio of *Atriplex* shrubs is based on estimates by Brewster (1968), while the estimate of four for the root:shoot ratio of *Artemisia* is similar to the estimates for cold desert shrublands in Jackson and others (1996). By revising the carbon stock data of Bradley and others (2006) to include these rough estimates, it can be demonstrated that the loss of belowground biomass carbon has the potential to contribute greatly to the effect of burning on carbon storage in these shrublands (figure 2). Using these estimates, the biomass carbon stocks in the salt desert shrubland were reduced eight-fold through burning and conversion to annual grasslands, while those of sagebrush steppe were reduced from at least six-fold to over fifty-fold.

It is true that belowground carbon from shrub roots is still present for some undetermined length of time post-conversion, after the large pulse of CO₂ emission from the combustion of the above-ground shrub biomass. But ultimately this carbon will be released to

the atmosphere, and without actively growing shrubs to replenish this belowground stock, the effect will be conversion of this formerly carbon-efficient system into a long-term source of atmospheric C. Estimates of biomass C loss from the study of Bradley and others (2006) ranged from 1.1 to 6.5 metric tons per hectare for aboveground biomass C, 8.6 to 26.4 metric tons per hectare for belowground biomass C, and 9.8 to 32.8 metric tons per hectare for total biomass C.

Bradley and others (2006) combined their estimates of the areal extent of conversion to cheatgrass monoculture in the Great Basin with their estimates of reduction in above-ground biomass C stocks as a consequence of this conversion to calculate total biomass C released to the atmosphere (table 2). They estimated that about 8 teragrams of C have been released to the atmosphere through shrubland conversion to annual grassland in the Great Basin as of 2006, and the potential for continuing type conversion and carbon release is immense. Adding estimated long-term belowground biomass carbon stock reduction resulted in an estimate of 29 to 60 teragrams of C that will ultimately be released to the atmosphere as a consequence of type conversion from shrubland to annual grassland that has already occurred in the Great Basin.

Invasive annual grass monocultures are not only very poor at carbon sequestration in terms of standing biomass relative to shrublands, but also tend to concentrate their SOC near the surface and to facilitate very rapid turnover of both soil C and N (Norton and others 2004). This is perhaps one reason why it has been difficult to demonstrate direct losses of SOC following annual grass invasion or conversion to annual grass dysclimax (Gill and Burke 1999, Ogle and others 2004, Bradley and others 2006). Most of these studies have examined only the near-surface

soil, where SOC under annual grasslands is concentrated. The technology for the study of deep SOC remains cumbersome, so that information on this fraction of the carbon pool is not readily obtained.

Shrubland Degradation and Windblown Dust

Another consequence of anthropogenic disturbance on a landscape scale in arid and semiarid regions is a large increase in the load of windblown dust. To examine the magnitude of this effect, Neff and others (2008) analyzed rates of sediment accumulation in mountain lakes in southwestern Colorado over the last 5000 years. They showed clearly that the rate of sediment accumulation peaked very sharply in the second half of the nineteenth century, a time frame that corresponds with a massive increase in the scope and intensity of livestock grazing in the arid and semiarid regions to the west. These workers further demonstrated using mineralogical analysis that these sediments were not of local origin, but instead represented deposits of windblown dust from the valleys to the west of the watershed.

Livestock grazing and other human activities that disturb the surface soils of deserts generate dust by removal of herbaceous plant cover and, often more importantly, through destruction of the cryptobiotic soil crust that stabilizes the surface in many desert regions (Neff and others 2005). These effects are further exacerbated by annual grass invasion and associated frequent fire. Annual grass cover provides some protection against wind erosion relative to bare ground, but it prevents cryptobiotic crust recovery, resulting in increased dust generation, especially when these areas burn. The Milford Flat fire of 2007 was the largest wildfire in the history of Utah (Miller and others 2011). An enduring legacy of this fire has

Table 2. Estimated biomass carbon loss as a consequence of conversion from cold desert shrubland to cheatgrass (*Bromus tectorum*) monocultures in the Great Basin as of 2006 (adapted from Bradley and others 2006).

	Salt Desert Shrubland	Sagebrush Steppe	Total
Aboveground biomass C loss (tons/ km ²)	110	250-650	360-670
Estimated total biomass C loss (tons/km ²)	1000	1500-3200	2500-4200
Estimated area burned (km ²)	2,000	18,000	20,000
Estimated aboveground biomass C loss (teragrams)	0.2	4.5-11.7	4.7-11.9
Estimated total biomass C loss (teragrams)	2	27-58	29-60

been massive dust storms that have swept windborne dust into the urban areas of northern Utah and onto mountain watersheds. In addition to direct impacts on air quality and human health, this windborne dust exacerbates the effects of climate change through its effect on snow melt rates.

Snow cover has the highest albedo (light reflecting ability) of any natural land surface, and this ability to reflect light also reduces heat loading and melting rate (Flanner and others 2009). When particulate matter, such as dust or carbonaceous pollutants, is deposited along with snow, it lowers the albedo of the remaining snow cover as the snow melts, because the dark particles are concentrated near the surface of the snow. While it is true that particulate matter in the air lowers insolation and heat load on snow at the surface, this 'dimming' effect is more than compensated by the reduction in snow albedo from these particles once they are deposited ('darkening effect'). This effect is especially pronounced in spring, when large areas are snow-covered and incident solar radiation is high. Flanner and others (2009) found that progressively earlier snow melt dates observed in Europe over the last few decades are almost as much due to this snow darkening effect of pollutants from fossil fuel combustion as to longterm increases in spring temperature caused by global warming. Moreover, the positive feedback from earlier snow melt caused by darkening created warmer spring temperatures independently of the effects of global warming, thus compounding the problem.

Though not as potent a darkening agent as carbonaceous pollutants, windborne dust can also significantly increase snow melt rates (Painter and others 2007). Spring dust storms in the desert region to the west of the mountain study area in southwestern Colorado resulted in several dust-on-snow deposition events per year, with more events in a drought year (2006, 8 events) than in an average moisture year (2005, 4 events). These dust-on-snow deposition events resulted in snow cover durations that were decreased by 18 to 35 days. Shortened snow cover duration has measurable ecological impacts at the local scale in alpine and subalpine areas (Steltzer and others 2009). More importantly, it also has the potential to significantly reduce water yields from mountain watersheds. Given that most of the agricultural and culinary water supplies in the Interior West are closely tied to mountain snowfall, and that the thickness and duration of the snow pack

and its rate of melting have a strong impact on the ability to harvest this water supply, the fact that desert dust storms can shorten the duration of snow cover in mountainous areas downwind by a month or more should be of grave management concern (Painter and others 2007).

Managing Desert Shrublands for Climate Change Mitigation

Climate change mitigation through desert shrubland management has the goal of maintaining or restoring adapted native shrubland vegetation that produces maximum carbon storage in the long term by exploiting all available niches and thereby maximizing productivity. It is likely that the vegetation that evolved in response to the selective forces in a particular environment will be best able to exploit its resources. This vegetation includes the woody shrub overstory, the herbaceous understory, and also the cryptobiotic crust community that occupies the interspaces. All these components are essential for longterm stability, including surface stability, and sustained carbon storage capacity.

An intact shrubland community is much more likely to be resilient in the face of continued climate change and other disturbances than 'shrub plantations' analogous to the tree plantations currently being proposed and implemented for carbon sequestration. Emphasizing shrubs to the exclusion of other community components in a short-sighted effort to maximize carbon storage would probably result in vegetation that would require intensive management to be sustained. Annual grass weed invasion of the bare interspaces and consequent shrub loss through fire would be a constant threat. A more realistic goal, and one that is bound to be more effective in the long term, is to manage for intact shrubland communities that can rebound even from disturbances such as prolonged drought and fire without high risk of conversion to annual grass dysclimax. Both prevention of further degradation and restoration of degraded shrublands are part of this management scenario.

Cold desert shrublands in the Interior West currently exist in one of three states along a continuum of ecological condition. Some sites still have relatively high-condition shrubland, with native understory and cryptobiotic crust still intact. Many more sites,

perhaps most of the area still occupied by shrubs, are in some intermediate condition, with native perennial understory and/or cryptobiotic crust damaged or absent and with annual weed invasion in the understory. These sites are often at high risk of conversion to the third state, which is loss of the shrub overstory through fire and post-burn dominance by annual grass weeds. Shrublands in these different states present different challenges and opportunities for management for carbon sequestration and windblown dust abatement.

Obviously, the most important consideration for high-condition shrublands is prevention of degradation. This means keeping the cryptobiotic crust and the herbaceous understory in the best possible condition. This minimizes the probability of massive annual grass expansion after fire and also maintains surface stability to minimize dust generation. Direct protection from invasion, for example, by controlling nearby weed infestations that could be propagule sources, is another way to maintain ecosystem integrity, as is providing priority protection in the event of wildfire. Even though occasional wildfire was a natural occurrence before settlement, especially in sagebrush steppe, protection from burning under current conditions is a top priority because of the threat of annual grass invasion.

Shrublands in intermediate condition often present more problems than opportunities in terms of improvement for climate change mitigation. Protecting from further disturbance may result in little improvement in these shrublands. Loss of the seed bank of native understory species limits recruitment, and the cryptobiotic crust often cannot recover because of the heavy litter resulting from annual grass invasion. In addition, a common occurrence, especially in sagebrush steppe, is shrub stand thickening or shrub canopy closure in response to loss of understory vegetation. The site at Jungo (Bradley and others, 2006) seems to represent such a scenario. Sagebrush standing biomass was very high, and the understory was completely dominated by cheatgrass. Such a site could be described as 'walking dead' in terms of the risk of conversion to annual grassland, as eventually a shrub-destroying fire is nearly inevitable. Natural shrub recovery after fire is often nil for dominant shrub species like

sagebrush and shadscale (*Atriplex confertifolia*), which cannot resprout after fire and rarely establish from seed in areas of high annual grass competition. Active management of shrublands with an understory dominated by cheatgrass will necessitate the development of effective tools to eliminate cheatgrass, reduce shrub cover if necessary, establish understory species, and encourage cryptobiotic crust recovery, all with a minimum of surface disturbance. At present such tools are largely unavailable.

Shrublands that have been converted to annual grass dysclimax communities have usually been given up for lost because of the futility of seeding into dense annual grass stands. But these annual grass dysclimax communities present the most hopeful scenario for increased carbon sequestration. If restoration of these communities is successful, substantial gains in carbon storage can be achieved. There should therefore be a strong emphasis on research aimed at increasing restoration success in areas that no longer support perennial vegetation. Many of the same tools needed for improving degraded shrublands will be needed for restoration of areas that no longer support shrubs, namely innovative methods for annual grass weed control, and new approaches to improving seeding success in environments with low and variable precipitation. At present most seedings in these environments fail, which may seem discouraging. But this points the way toward the development of new approaches that, while they may be more expensive up front, could result in greatly improved seeding success and therefore a much better cost: benefit ratio for shrubland restoration in the long run. It is our challenge as researchers to develop these new approaches. With climate change mitigation as the goal, rather than management of these shrublands for consumptive uses such as livestock grazing, the most creative scientists among us will be inspired to 'think outside the box' and devise the methodology needed to make Interior Western shrublands a significant carbon sink. Even better, along with our partners in management, we will at the same time have the opportunity to enhance the many other ecosystem services and amenity resources provided by these landscapes.

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Land-Use Legacies of Cultivation in Shrublands: Ghosts in the Ecosystem

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ABSTRACT

Shrublands across the West are currently threatened by land uses such as urban sprawl, energy development, and agricultural development which impact ecosystem function through altered fire cycles, expansion of invasive species, modified hydrology, and intensified soil erosion. Historically, shrubland ecosystems have already been impacted by many of these same disturbances. Unlike our forested ecosystems, much of the land-use history in our shrublands has been forgotten or ignored. But our human endeavor can leave lasting changes on the landscape, referred to as “land-use legacies”, for decades to centuries. Looking for land-use legacies does not equate with looking for someone to blame. People have always sought to use the resources from the land on which they live. By not recognizing land-use legacies, however, we are not taking full advantage of the potential to learn about how shrublands respond to and recover from a myriad of disturbances. This paper will highlight one of the overlooked land uses within shrublands associated with homesteading - cultivation. Understanding what has happened on the landscape in the past can offer a great deal of information regarding its potential in the future.

In Monaco, T.A. et al. comps. 2011. Proceedings – Threats to Shrubland Ecosystem Integrity; 2010 May 18-20; Logan, UT. Natural Resources and Environmental Issues, Volume XVII. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan Utah, USA.

INTRODUCTION

Historic land uses can leave lasting impacts on ecosystems, known as “land-use legacies”, for decades to centuries (Foster et al. 2003). However, evidence of historic land use is not always visible on the landscape. In addition, some historic land uses are eclipsed by the attention that other uses receive, such as livestock grazing. One of these “ghosts” in the ecosystem that is not always easy to see and is often overlooked is homesteading. Homesteading is often forgotten because the material evidence of this land use has been disappearing over time (figure 1). Therefore, without records of what happened or knowledge of what to look for, it would be easy to miss the fact that people had, at one time, homesteaded in an area. But just because the material evidence is not visible does not mean the land use associated with homesteading has not left a legacy. This paper will highlight one of these often overlooked land-use legacies - cultivation.

HOMESTEADING AND CULTIVATION

The Homestead Act of 1862 allowed for acquisition of up to 160 acres of federal land. This legislation

required that the applicant be a head of household or 21 years of age and either be a citizen of the United States or provide proof of declaration to become one. To gain patent (or “prove up”) on the claim, applicants were required to prove five years residence and cultivation of the land. This process was designed to show that the patentee intended to live on the claim and would add value to it through investment in infrastructure such as fencing, water developments, permanent structures and cultivation (Gates 1968).

Cultivation, along with livestock grazing, was a primary land use during homesteading. Although the Homestead Act of 1862 required proof of cultivation, it was not until the Enlarged Homestead Act of 1909 that legislation required a certain amount of land be cultivated within a specified timeframe in order to gain patent (Peffer 1972). The Enlarged Homestead Act doubled the acreage of land available for patent to 320 acres. Under this new law, 20 acres had to be under cultivation by the second year and 40 acres continuously under cultivation from the third year to the final year (Peffer 1972). This new cultivation requirement was a product of the popularity and promotion of dry farming (agriculture without irrigation) in the U.S. (Gates 1968; Peffer 1972).

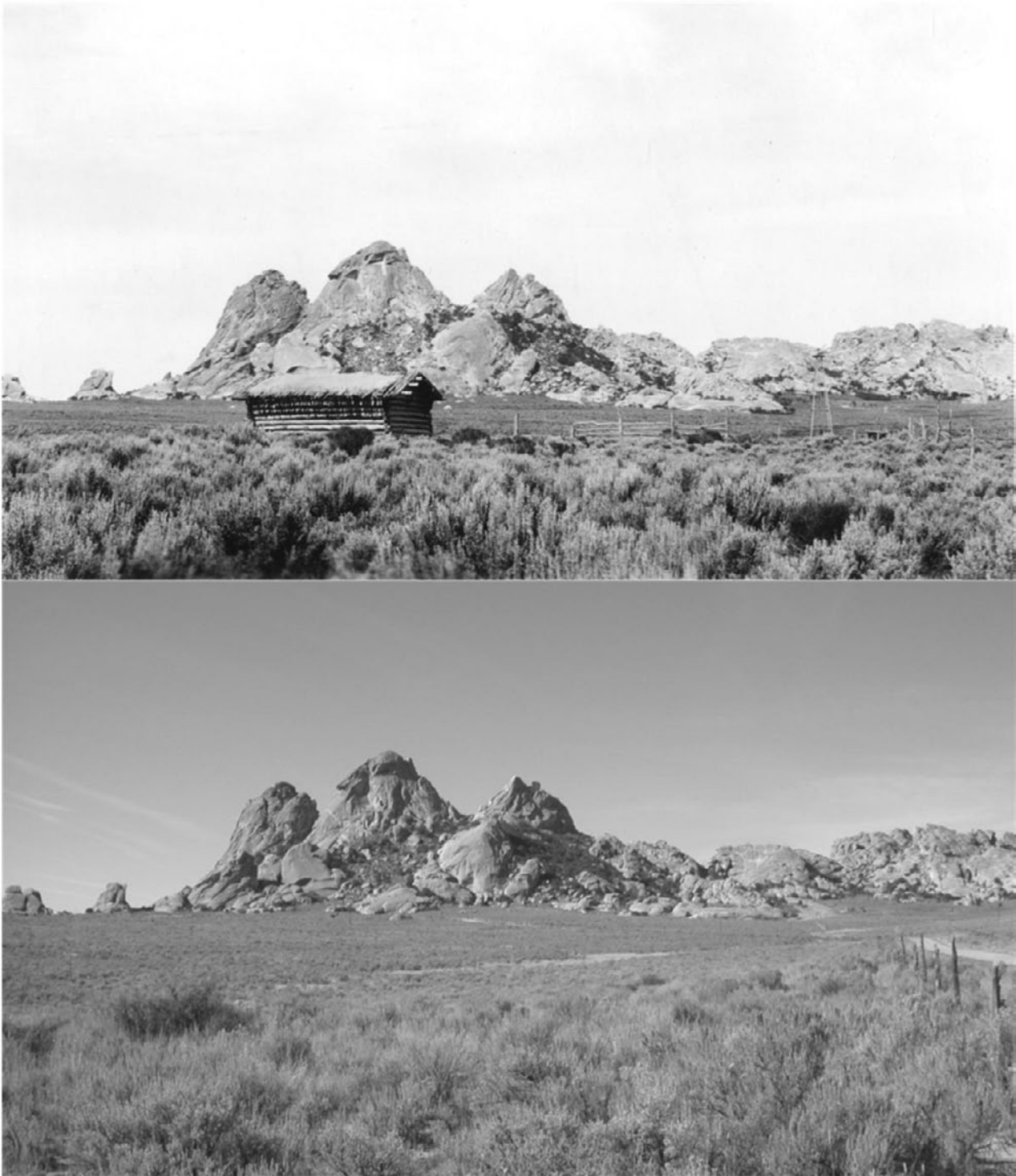


Figure 1. The material evidence of homesteads can fade over time, but the land-use legacies of cultivation remain. The top photo shows structural remains of a homestead in southern Idaho in 1930 (Photo courtesy of Utah State Historical Society). The bottom photo shows the same area in 2005 (Photo by Lesley Morris).

Dry farming methods at the time were straight forward but very labor intensive. First, the land had to be cleared of shrubs and other vegetation. This was accomplished in a variety of ways including dragging a rail or a railroad tie behind a team of horses or

digging them out with an axe and hoe (Scofield 1907; Schillinger and Papendick 2008). Once cleared, the land was plowed as “deep as possible” to break up the soil, usually around 7 to 10 inches in depth (Buffum 1909). Finally, the field was “harrowed” with a

wide frame fixed with large spikes hanging toward the ground (Schillinger and Papendick 2008). Harrowing was used to pulverize the soil surface and break any capillary action which might allow water to evaporate (figure 2; Scofield 1907; Schillinger and Papendick 2008). Half of the field was kept in this harrowed state for a season to accumulate and “store” water while the other half was planted (Buffum 1909; Pepper 1972). The idea was that if no other plants were allowed to use the soil moisture, all of it would be available to the crop planted on the site. Thereby, dry farming only used water stored in the soil from precipitation without additional irrigation.



Figure 2. A dry-farm field ready for planting in Park Valley, Utah in 1911 (Photo courtesy of Utah State Historical Society).

Several factors drove the popularity of dry farming. It was called the “new science of agriculture” because of the research focus it gained at the agricultural universities in the West (Morris et al. 2011a). It was promoted by railroad companies because they could advance the use of their tracks as transport to markets as well as sell off their most arid land grants from the federal government (Strom 2003; Orsi 2005). Land companies purchased railroad land grants and went into business promoting the development of arid lands for agriculture (Bowen 2003; Morris et al. 2011a; Wrobel 2002). Dry farming, particularly that of dry-land wheat, was also promoted by the federal government through legislation that subsidized wheat prices during World War I and through legislation like the Enlarged Homestead Act. The combination of promotion, legislation and economics made the Enlarged Homestead Act the most popular of all the federal provisions to dispose of the public lands in the West. In the first year of its passage, applications for patents were filed on over 18 million acres of land (Gates 1968) and the following decade had the most homesteads filed.

Starting in the 1920s, several factors began to unravel dryland farming in the West. First, the price of wheat, which had been subsidized by the federal government during World War I, declined rapidly (Hyde 1937). Secondly, many blamed the droughts beginning in the 1920s and continuing through the 1930s for crop failures (Bowen 2001; Gates 1968). However, the drought years simply made a bad situation worse because many of the locations where dry farming was attempted were unsuitable from the start (Roet 1985). In the rush created by land companies to gain land and grow wheat, many settlers were lured to submarginal lands where agriculture of any kind could not thrive due to low precipitation, harsh climate, and unsuitable soils (Bowen 2001; Bowen 2003; Wrobel 2002). Areas that were less suitable for agriculture from the beginning have an even greater capacity for cultivation legacies (Cramer et al. 2008). Though many of these abandoned farms no longer have structures on them to indicate this historic land use, the legacies of dry farming remain on the landscape. Often, abandoned old fields can be seen from aerial photographs for decades to almost a century after they were first cultivated (figure 3; Elmore et al. 2006; Morris and Monaco 2010; Stylinski and Allen 1999).

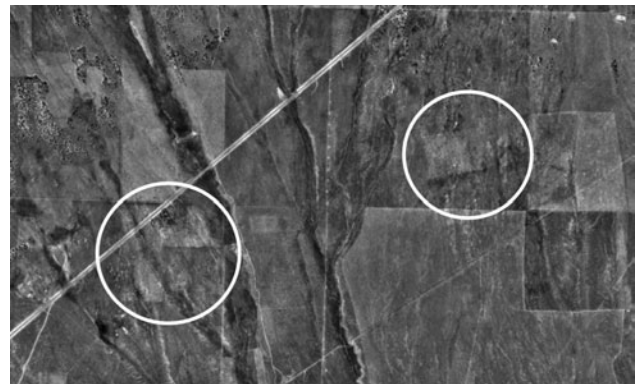


Figure 3. Aerial photo taken in 1999 showing two old fields (in circled areas) that were first cultivated nearly a century ago then abandoned (Photo courtesy of USGS).

LAND-USE LEGACIES OF CULTIVATION

Cultivation leaves legacies on shrubland vegetation, hydrology and soils. Native species recovery after cultivated lands are abandoned may take decades (Daubenmire 1975; Rickard and Sauer 1982; Standish et al. 2007) to over half a century (Elmore et al. 2006; Morris et al. 2011b; Simmons and Rickard 2002; Stylinski and Allen 1999). Old fields can have lower total plant cover, lower species richness, and

lower frequency and cover of perennial grasses (Elmore et al. 2006). In addition, forb cover is generally lower in old fields (Dormaar and Smoliak 1985; Morris et al. 2011b; Rickard and Sauer 1982; Simmons and Rickard 2002;) while exotic forb cover is higher (Morris et al. 2011b; Rickard and Sauer 1982; Stylinski and Allen 1999). Old fields also tend to be dominated by invasive grasses, such as cheatgrass (*Bromus tectorum* L.) (Daubenmire 1975; Elmore et al. 2006; Rickard and Sauer 1982). Shrub composition can be altered in old fields and recovery of sagebrush cover after dry farming can take longer than other disturbances, well over 90 years in some places (Morris et al. 2011b). Seed banks of native species tend to be impoverished by cultivation (Cramer et al. 2008) while agricultural weeds form persistent soil seed banks that are likely to also dominate the soil seed bank after abandonment (Ellery and Chapman 2000; Cramer and Hobbs 2007).

The land-use legacies of cultivation also impact hydrology including soil moisture, soil water holding capacity, run off and infiltration. Cultivation legacies can have a greater effect on differences in soil water movement between plowed and never plowed sites than the differences in soil water movement between two soil series (Schwartz et al. 2003). In fact, soil hydraulic conductivity can remain affected for well over 25 years after cultivation ceases and such alterations may be very difficult to restore (Fuentes et al. 2004). Water availability can also be reduced by soil compaction in old fields (Standish et al. 2006). Finally, plowing has been shown to reduce infiltration rates (Gifford 1972) and the recovery potential of infiltration rates on plowed land with grazing is much lower than is predicted for grazing alone (Gifford 1982).

Cultivation legacies impact the physical and chemical properties of soils (Standish et al. 2008). Physical changes, such as soil compaction can create physical boundaries to plant development (Buschbacher et al. 1988; Uhl et al. 1988; Unger and Kaspar 1994) or soil loosening which can favor invasive species (Kyle et al. 2007). The physical disturbance of soil through cultivation increases the potential for erosion (Navas et al. 1997; Schillinger and Papendick 2008). There are also legacies that manifest as changes in soil organic carbon and fertility (McClachlan 2006). Loss of soil organic matter content in cultivated land was reported at 20-25 percent in comparison to

noncultivated adjacent land within the first 30 years of dry farming (Bracken and Greaves 1941, Schillinger and Papendick 2008). Total soil organic matter can be lower in old fields up to 53 years after abandonment even while rebuilding at smaller scales under plants (Burke et al. 1995). However, even when systems regained some soil organic matter, the rate of recovery had not matched the rate of loss during cultivation (Ihori et al. 1995).

WHY DO THESE LAND-USE LEGACIES MATTER?

Homesteading for the purpose of dry farming was widespread across the West and, therefore, so was the abandonment of this land use. It was estimated that nearly 23 million acres of rangeland were cultivated and abandoned by the late 1930s (Stewart 1938). In the Intermountain West, one fourth of the 12 million acres of degraded rangelands were reportedly abandoned plowed lands (Pearse and Hull 1943). There were 2 million acres of abandoned dry farmed and irrigated land in southern Idaho alone by 1949 (Stewart and Hull 1949). Land-use legacies resulting from cultivation now exist in all landownership types including private property and public lands managed by the Bureau of Land Management, National Park Service, and the US Forest Service. Therefore, the legacies in these old fields have the potential to underlie all management objectives. Old fields from homesteading may exist within rangeland seedings on private property or within areas slated for restoration to enhance recreation and wildlife use. They can be part of areas where fuels management is needed or revegetation is desired following wildfires. Better knowledge of the "ghosts" of land-use past in shrublands, like cultivation, will provide more understanding of the function of these systems and reduce the likelihood of misunderstanding their future potential (Foster et al. 2003).

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Land Use and Habitat Conditions Across the Southwestern Wyoming Sagebrush Steppe: Development Impacts, Management Effectiveness and the Distribution of Invasive Plants

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ABSTRACT

*For the past several years, USGS has taken a multi-faceted approach to investigating the condition and trends in sagebrush steppe ecosystems. This recent effort builds upon decades of work in semi-arid ecosystems providing a specific, applied focus on the cumulative impacts of expanding human activities across these landscapes. Here, we discuss several on-going projects contributing to these efforts: (1) mapping and monitoring the distribution and condition of shrub steppe communities with local detail at a regional scale, (2) assessing the relationships between specific, land-use features (for example, roads, transmission lines, industrial pads) and invasive plants, including their potential (environmentally defined) distribution across the region, and (3) monitoring the effects of habitat treatments on the ecosystem, including wildlife use and invasive plant abundance. This research is focused on the northern sagebrush steppe, primarily in Wyoming, but also extending into Montana, Colorado, Utah and Idaho. The study area includes a range of sagebrush types (including, *Artemisia tridentata* ssp. *tridentata*, *Artemisia tridentata* ssp. *wyomingensis*, *Artemisia tridentata* ssp. *vaseyana*, *Artemisia nova*) and other semi-arid shrubland types (for example, *Sarcobatus vermiculatus*, *Atriplex confertifolia*, *Atriplex gardneri*), impacted by extensive interface between steppe ecosystems and industrial energy activities resulting in a revealing multiple-variable analysis. We use a combination of remote sensing (AWiFS (¹ Any reference to platforms, data sources, equipment, software, patented or trade-marked methods is for information purposes only. It does not represent endorsement of the U.S.D.I., U.S.G.S. or the authors), *Landsat* and *Quickbird* platforms), *Geographic Information System (GIS)* design and data management, and field-based, replicated sampling to generate multiple scales of data representing the distribution of shrub communities for the habitat inventory. Invasive plant sampling focused on the interaction between human infrastructure and weedy plant distributions in southwestern Wyoming, while also capturing spatial variability associated with growing conditions and management across the region. In a separate but linked study, we also sampled native and invasive composition of recent and historic habitat treatments. Here, we summarize findings of this ongoing work, highlighting patterns and relationships between vegetation (native and invasive), land cover, landform, and land-use patterns in the sagebrush steppe.*

In Monaco, T.A. et al. comps. 2011. Proceedings – Threats to Shrubland Ecosystem Integrity; 2010 May 18-20; Logan, UT. Natural Resources and Environmental Issues, Volume XVII. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan Utah, USA.

INTRODUCTION

Beginning in 2005, a multi-partner, long-term, science and management cooperative, the Wyoming Landscape Conservation Initiative, was created to coordinate efforts of public and private land managers across a vast and heterogeneous landscape. The

U.S. Geological Survey, building on a foundation of several overlapping but uncoordinated programs of research and management across the region, is working to assess, monitor, and enhance ecological understanding of aquatic and terrestrial habitats across southwestern Wyoming. Here, we discuss the results and implications of three projects aimed at

vegetation distribution and conditions across the region. This includes building an understanding of the distribution and condition of sagebrush habitats across this large and heterogeneous landscape, including mapping of dominant vegetation and weed distributions and assessment of the role of management treatments in distribution of native vegetation, weeds and wildlife.

A foundational component of this research has been the development and implementation of multiple-scale mapping of plant cover without using type classifications. By using a combination of field collections nested within three scales of remote sensing data (QuickBird, 2.5 meter resolution, Landsat, 30 meter resolution, and AWiFS, 56 meter resolution), we developed the connections between surface patterns and spectral responses to estimate cover for a suite of eight soil and vegetation classes. Initiated for the WLCI (Wyoming Landscape Conservation Initiative), this effort began with a sub-state region, expanded to include all of Wyoming and it is now being applied across the sagebrush steppe. This information forms the most comprehensive remote sensing based assessment of sagebrush communities to date. Following tests of accuracy, change detection and repeatability, these methods, used to determine the current status, may be adopted as the core of monitoring the distribution and condition of shrub steppe communities. Importantly, for the current assessment, and for subsequent monitoring, these methods provide locally relevant detail (30 m resolution) at a regional scale (state-wide and larger).

With a clarified picture of the distribution of sagebrush communities across the study region, we remain faced with questions about the condition and productivity of these ecosystems. To begin to address these questions, we estimated the distribution of two landscape-scale drivers of change within natural and managed areas: biotic invasions induced by land use and management activities that intentionally altered habitat conditions. We assessed the relationships between specific, land-use features (for example, roads, transmission lines, industrial pads) and invasive plants, including their potential (environmentally defined) distribution across the region as an indicator of the extent of anthropogenic influences beyond the footprint of roads, urban and exurban domestic developments, agricultural fields, and energy infrastructure (oil, gas, and coal-bed methane). This required an accurate depiction of the

distribution of these surface disturbances (land-use conversions, industrial sites, treatment locations), however these data were not consistently available, therefore, a large part of this process has been development of accurate representation of human activities across the landscape. Beyond delineation, our research focus is the biotic implication of these features within and beyond their boundaries.

While major shifts in land use may be tracked through mapping and monitoring the distribution of human infrastructure (for example, roads, zoning, urban areas), the widespread, long-term practice of conducting habitat treatments by land management agencies has been untracked, poorly documented, and the impacts have not been well assessed. While individual treatments may be small (in areal extent), many are not, and the accumulation of treated areas across the landscape since initiation (circa 1940s) can be locally significant. Furthermore, understanding potential benefits and risks associated with particular treatment techniques is needed for adaptive management. Based on this need, we were able to use recently developed information (Wyoming Wildlife Consultants, LLC, unpublished data) to identify and locate historic treatments in southwest Wyoming, which we began sampling in 2010 (vegetation cover and composition). Wyoming Wildlife Consultants conducted parallel studies of wildlife use of these treated areas. The objective of this on-going work is to determine the long-term, persistent effects of these habitat treatments, especially the effects of habitat treatments on the ecosystem, including wildlife use and native and invasive plant abundance and structure.

STUDY AREA

The focal region for our research included over 7.7 million hectares (19 million acres) with variable environmental and land-use patterns including Green River and Great Divide Basins and several adjacent, smaller basins (figure 1). In addition, due to interest of land managers, the sagebrush mapping project was extended beyond these initial boundaries across the State of Wyoming. The research and management interests discussed here focus on the northern sagebrush steppe, primarily in Wyoming, but the potential implications and applications of these results may be extended into similar areas of Montana, Colorado, Utah and Idaho. The study area included a range of sagebrush types typical of northern, shrub-steppe. A majority of the region was dominated by

Wyoming sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) interspersed with salt-flats dominated by greasewood (*Sarcobatus vermiculatus*) and saltbush (*Atriplex gardnerii*) and varying abundances of rabbitbrush (primarily *Chrysothamnus viscidiflorus*). Throughout the region, native bunchgrasses such as bluebunch wheatgrass (*Pseudoroegneria spicata*) and needlegrass (*Achnatherum contractum*, *A. hymenoides*) mix with native and introduced wheatgrasses, including crested wheatgrass (*Agropyron cristatum* var. *cristatum* *A. cristatum* var. *desertorum*), bottlebrush squirreltail (*Elymus elymoides*), and western wheatgrass (*Pascopyrum smithii*). Importantly, there was also a wide-spread but heterogeneous distribution of annual, biennial, and perennial weedy plants including annual bromes (*Bromus tectorum*, *B. arvensis*), desert alyssum (*Alyssum desertorum*), halogeton (*Halogeton glomeratus*), Russian thistle (*Salsola tragus*), and tumble mustard (*Sisymbrium altissimum*). Invasive plants can alter the composition, productivity and forage quality of the ecosystem, making the distribution of these species, both across the range and within specific treatments, important for assessing and managing habitat conditions.

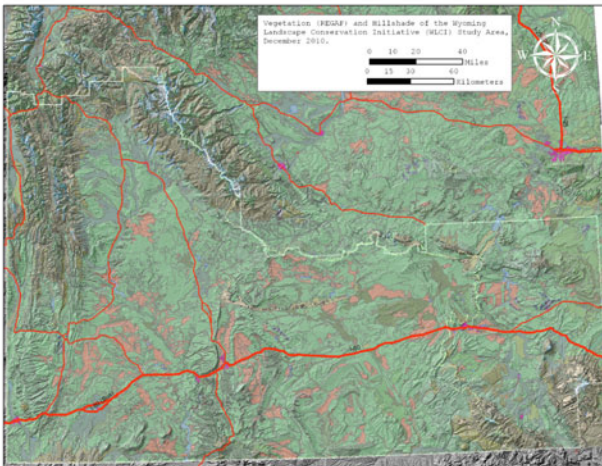


Figure 1. The Wyoming Landscape Conservation Initiative Area and the State of Wyoming, U.S.A. Shades of green and beige represent dominant vegetation types. Bright green is sagebrush steppe (dominates the scene); dark beige areas within the sagebrush steppe are more arid, desert shrub and saltbush flats. Light beige, within the central basin, represents active sand-dunes. Foothills woodlands are represented by olive, and are also recognizable by topographic relief depicted in the underlying topographic hillshade, with higher elevation forests appearing in dark green above the band of foothills. Red-lines represent major highways.

The climate is dry continental, with mean annual precipitation totals of 10 to 13 inches being typical (Western Regional Climate Center, www.wrcc.dri.edu) For much of the region mean maximum temperatures in July range from 85° to 95°F, with mean minimum in January typically between 5° and 10°F (ibid). Our samples are distributed across heterogeneity in soils, geology, topography, climate, hydrology, and dominant vegetation in addition to differences in land-use attributes that were targeted by design.

This region has historically supported (circa 1900) agricultural and natural resource extraction economies. Despite concerns about the welfare of wildlife and ecosystems, increasing energy demand and expanding infrastructure results in continuing impact by extensive, and often intensive, industrial energy activities. Thus, modern disturbances and landscape fragmentation are being superimposed on a long-history of land-use impacts. Understanding the current interactions of naturally determined and anthropogenically influenced environmental conditions is critical for successful conservation, restoration and management of these semi-arid landscapes.

Multi-Scale Sagebrush Mapping And Resource Inventory

METHODS

We developed methods to combine three scales of satellite imagery (2.4-m QuickBird, 30-m Landsat TM, and 56-m AWiFS) using limited but rigorous and directed ground sampling to produce continuous predictions for eight sagebrush steppe vegetation components across the state of Wyoming.

High resolution QuickBird (QB) images each covering 64 km² were segmented into patches to distribute field sampling sites across polygons representing spectral variations in the target area. Each image was also classified into 30 unsupervised classes, and the majority class in each segmented polygon was determined. To correlate surface conditions with remotely detected variability across the image, we systematically sampled polygons in each spectral class. Typically two polygons were sampled from each majority class, for a minimum of 60 sampling locations per QB footprint.

The composition of vegetation, bare ground and litter in each polygon were assessed using ocular estimation of 1-m² quadrats. Fourteen (14) quadrats were divided evenly (5m apart) along two 30-m transects (7 per transect; figure 2); these values were averaged to define the cover of the site. Transects were aligned parallel, but offset (creating a parallelogram footprint) with a maximum of 20 m separation between transects. These sample units were distributed across the 64 km² footprint, with replicates, to develop field data to represent spectral variability across the scene. Canopy cover of vegetation was estimated in 5 percent increments based on a conceptual “similar-to-satellite” interpretation, such that only the top-most layer of cover was recorded and the sum of all primary cover components could not exceed 100 percent. Shrubs and trees (if present) were identified to the species level, with sagebrush (*Artemisia* spp.) further distinguished to the subspecies level. Heights of shrub and tree species were estimated based on measurement of the tallest green vegetation (excluding seed stalks) of each species within each quadrat.

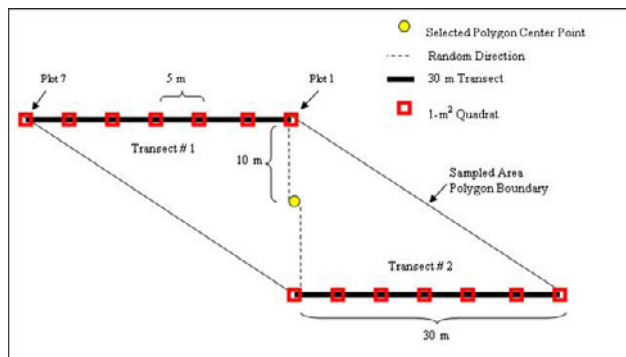


Figure 2. Physical layout of replicated field plots used to develop cover estimates for training Quickbird spectral signatures. This array was replicated within each unique spectral group (number per scene varies due to heterogeneity) within each targeted Quickbird scene (8km x 8km).

To apply the field data to the remotely sensed imagery, we defined sampled areas as the polygon created by connecting the start and end points of both transects at each location. For each component we calculated the mean value across the 14 quadrats, and these mean values were assigned to all QB pixels falling within a sampled area.

Using regression tree analysis to identify empirical relations between the component values and the QB

data (typically all four 2.4-m spectral bands and three additional bands of ratio indices), we classified the proportion of each of the components occurring within each entire QB image on a per-pixel basis. These per-pixel QB predictions were then resampled to 30-m Landsat and 56-m AWiFS pixels to provide the component training data for the model predictions at these larger scales. A number of additional data layers (image band ratios, ratio differences between image dates, ancillary topographic data) were also provided to the regression tree for model building.

RESULTS AND DISCUSSION

We produced continuous predictions for eight sagebrush steppe vegetation components across the state of Wyoming using three spatial scales of remotely sensed imagery. The four primary components were percent bare ground, percent herbaceous (grass and forb), percent litter, and percent shrub, which taken together represent 100 percent of all cover in a tree-less environment. The four secondary components include three subsets of percent shrub, including all sagebrush (*Artemisia* spp.), all big sagebrush (*A. tridentata*) subspecies, and only Wyoming sagebrush, as well as mean shrub height. Predictions revealed that bare ground had the most even distribution across the entire range; this is not surprising on this semi-arid landscape. Herbaceous vegetation and litter cover exhibited similarly broad ranges and distributions, especially when compared to shrub cover which is less uniform. Wyoming sagebrush had the most limited range of the variables we modeled.

Prediction accuracy varied by imagery type, image, and component. We used, root mean square error (RMSE, in the units of the component prediction) a useful measure of model accuracy to compare results. At the QB level, RMSE values ranged from 4.76 for sagebrush to 10.16 for bare ground, with 7.95 for shrub height. Accuracy at the Landsat and AWiFS scales were generally more variable than at the QB scale. Landsat RMSE values ranged from 5.46 for sagebrush to 15.54 for bare ground, with 11.2 for shrub height. AWiFS RMSE values ranged from 6.11 for sagebrush to 16.14 for bare ground, with 10.18 for shrub height.

We found that our component predictions outperformed those generated by LANDFIRE (Rollins and others 2006), the only comparable large-area

product. For the shrub component the RMSE of our model prediction was 6.04, as compared to 12.64 for LANDFIRE, and for herbaceous the RMSE was 12.89, versus 14.63 for LANDFIRE.

We believe our Landsat and AWiFS predictions provided enough detail for local application, span areas broad enough for ecosystem analysis, and provide a quantitative and repeatable framework for future monitoring. Research applying our component estimates to current and historical vegetation change, climate variation, sage grouse habitat distribution, and grazing trends are currently underway.

Land Use And Invasive Plants

METHODS

We developed data for species distributions using a sample of 123 sites distributed across the landscape, representing several ecological types and multiple land-use features. An important value created by the spatial modeling approach is leveraging the information contained in expensive field samples by projecting distribution estimates beyond sample sites. Here, we minimized the negative effects of projecting onto unsampled landscapes by including our sampled area within the projected area, thereby reducing the assumptions and errors associated with extrapolation to unsampled climate and landscape associations (Rodder and Lotters 2010). We developed a stratified-random sample design using a spatially explicit representation of anthropogenic features distributed across the WLCI study area (7.7 million hectares), which also captured important environmental variability by crossing geologic and soil types, precipitation and temperature gradients, and various topographic patterns.

We sampled paired, 1000m-long by 1m-wide belt-transects that were extended perpendicular to the margin of a target feature (in all cases except “control” sites); these were generally extended in divergent or opposite directions to capture community and species diversity across the site. Each 1m² was examined and all identifiable invasive plants were recorded, confirming the presence or absence of 30 species identified in county, federal and state noxious weed lists. We post-processed sites to add attributes representing environmental characteristics in a GIS (geographic information system, ESRI ArcMap 9.3) by associating sample locations with existing information

(for example, surface geology, dominant vegetation and road density). This combination allowed subsequent analyses including these variables as covariates of weed abundance. Based on observed distributions of species, we were forced to immediately revise our initial hypothesis that all species would show a linear or curvilinear decreasing relationship with increasing distance, because simple graphs demonstrated otherwise, for some species. We tested linear and log-linear transformed distance as predictors of species abundance using generalized linear models (R Development Core Team 2010) and discovered nearly ubiquitous, significant relationship between plot distance [increasing distance from anthropogenic features; $p < 0.05$ in all cases except log-linear for halogeton and linear for perennial pepperweed (*Lepidium perfoliatum*) which were not significant.] However, we also tested the contribution of potential environmental predictors, and discovered that the model fit was improved by adding an environmental covariate in all cases; this was generally the dominant surface geology or vegetation type.

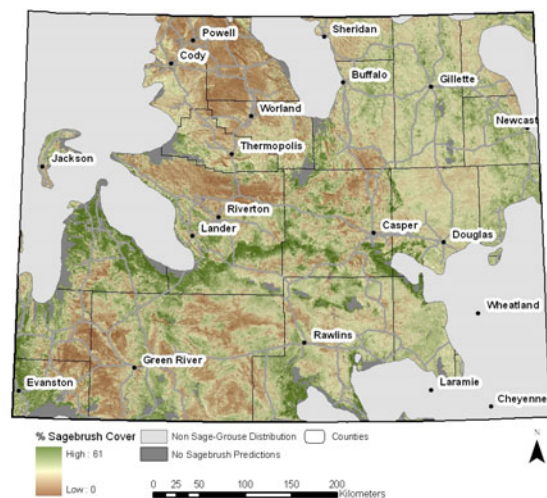


Figure 3. Continuous prediction of sagebrush cover (all species and subspecies combined) in Wyoming, U.S.A.

RESULTS AND DISCUSSION

We found clear connections between the distribution of several, prominent invasive plants and widespread rural land-use features including all classes of roads (highways, major and minor unpaved thoroughfares, spurs and driveways and double-tracks), active and reclaimed well-pads, pipelines and transmission lines.

We found the greatest richness of invasive plants associated with informal roads (double-track, two-track; figure 3) which likely receive variable, seasonal use, but little to no weed management. Active well-pads (oil, natural gas, and/or coal-bed methane), pipelines, and primary (county roads and similar, thoroughfares) and tertiary (short gravel spurs, driveways and dead-ends) roads contained a greater richness than the ambient conditions estimated by Control sites (figure 4). It is important to note that our "Control" sites do not offer unbiased, undisturbed data for comparisons. These sites were located more than 1000m, continually along their entire length, from any neighboring anthropogenic features, but they were embedded within utilized landscapes. Therefore, the data from these sites offered a basis for relative assessment of specific features as well as evidence of the wide-distribution of invasive plants.

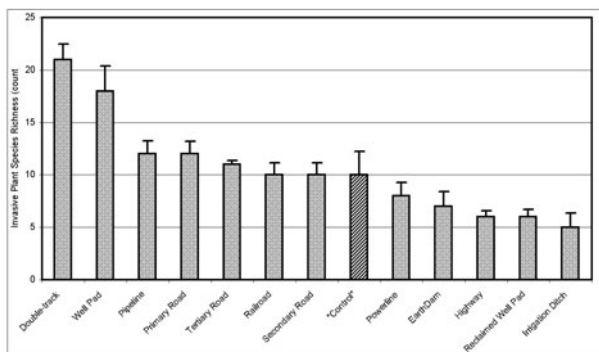


Figure 4. Observed richness (species count) of invasive plants relative to anthropogenic features within a rural, southwestern Wyoming landscape. Control sites were located more than 1000 m from the nearest anthropogenic feature; however these are clearly not "weed free" controls. These sites were surrounded by various intensities of land-use (especially roads and well pads), so rather than a true control, these sites document the "background" levels of invasion across the "untrammeled" landscape.

Although many species were not found in sufficient abundance, within our sampling design, to model individual feature-distance relationships, analysis of several abundant, recurring species reveals important patterns and distinctions in their local distributions. Generalized linear models revealed a significant, inverse relationship between distance (and log-linear transformed distance) from a given feature and abundance of cheatgrass, halogeton, perennial pepperweed, flaxweed, desert alyssum and Russian

thistle ($P > F$, 0.0000001, 0.0271, 0.0441, 0.000007, 0.0001, .0001, respectively). However, the abundance of weeds, taken in sum, did not decline with increasing distance ($P > F$, 0.3276) indicating the widespread abundance of weeds across many parts of this landscape. Weedy plants adjacent to major roads (primary roads) displayed the anticipated exponential decay curve (figure 5a) with the greatest abundance of invaders falling within 200 meters of the road and measurable abundance approaching zero near 400 meters. The distribution of weeds associated with secondary roads (large unpaved routes), tertiary roads and informal roads precluded fitting linear or curvilinear trends due to distance effect (figure 5). Thus, while some species did appear to decline in abundance between 400-600 meters away from targeted features, the expanse of invasion extended well beyond these distances, with little to no sign of decline. Of particular concern for managers in this region are annual bromes, especially cheatgrass (*Bromus tectorum*; also known as downy brome).

Cheatgrass has come to dominate vast, formerly sagebrush dominated, landscapes in neighboring regions, such as the Great Basin (Knapp 1996; Chambers and others 2007), making the species a major management concern across the Intermountain basins and northern steppe (Monsen and Shaw 2000). We found a wide distribution of cheatgrass in southwestern Wyoming, but it is not clear that the distribution of infrastructure is having an effect on these distributions, because although it exhibited a significant distance relationship, cheatgrass was observed in large abundances beyond 500m from the nearest feature. Our samples disclosed recognizable abundance of occurrences near features, and demonstrate decreasing abundance with increasing distance, as anticipated, when considering interactions with a single feature, such as informal, two-track roads (figure 6). However, in many cases, other road classes for example, weed occurrence is sustained at a distance greater than 500m from the nearest anthropogenic features (figure 6). This suggests that another, widespread environmental condition or activity is also responsible for driving the patterns of cheatgrass distribution and dominance in this region. Ongoing research is aimed at discerning the important driving factors for predicting, and restricting, the distribution of invasive plants relative to a combination of environmental factors.

Composition In Historic Habitat Treatments

BACKGROUND AND METHODS

Federal and state agencies and nongovernmental organizations have been funding habitat treatments across southwestern Wyoming for many years. There is a general recognition that monitoring of past and current habitat treatments have lacked designs and standardized approaches necessary for summarizing the effectiveness of current and past habitat treatments across spatial and temporal scales (Hughes and others 2000; Connelly and others 2004). Monitoring of restoration and habitat treatments is essential to determine their performance in order to make improvements and develop best management

practices to help guide the design and development of future habitat treatments and to improve the ability of these treatments to meet landscape conservation objectives locally, and across the landscape. To accomplish this multi-scale goal, we included field measurement of vegetation, soil and wildlife use (as indicated by fecal deposits), with remote sensing approaches for estimating plant productivity and phenology. Within this region, interactions between Greater Sage-Grouse (*Centrocercus urophasianus*; hereafter referred to as sage-grouse) and habitat conditions are critical for management planning, therefore direct estimates of wildlife response to treated habitats and developed and reclaimed habitats will inform adaptive management of wildlife habitats.

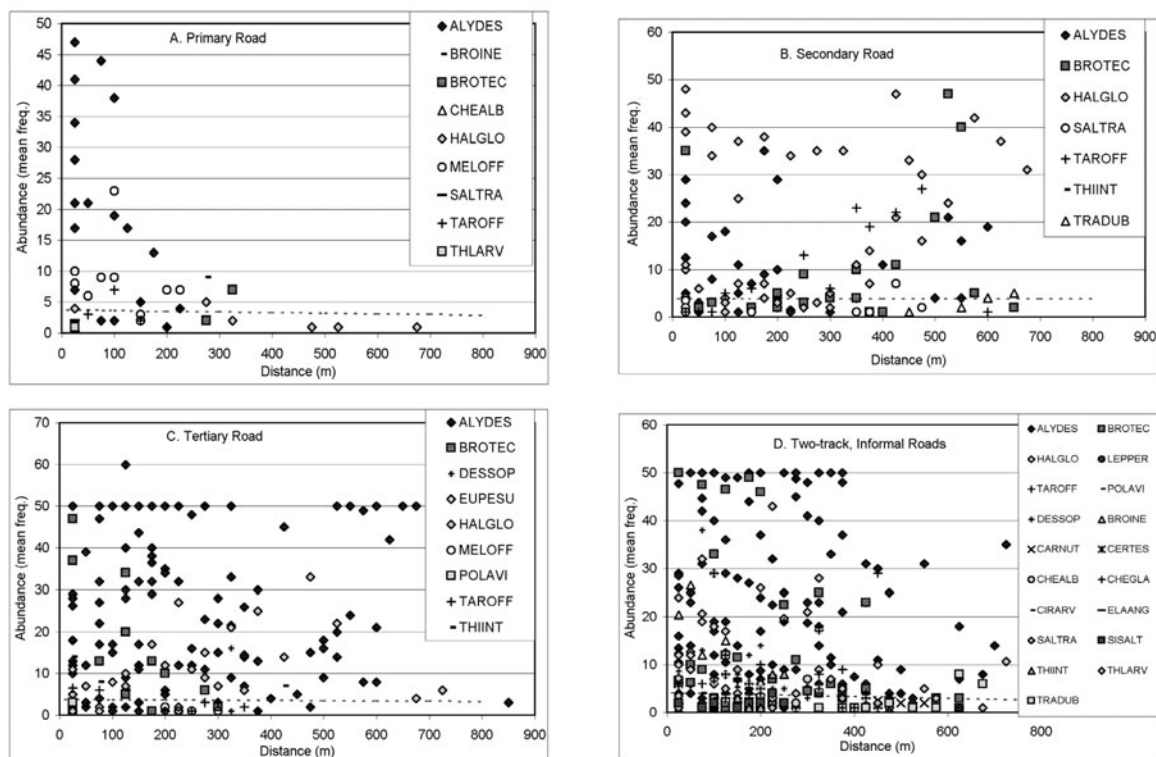


Figure 5 (a-d). Simple distributions of invasive plants observed in proximity to four (4) different sized road classes in southwestern Wyoming, U.S.A. The x-axis depicts the distance from a target feature based on observation of each 1m², aggregated into 25 m segments for each abundance estimate. Species abbreviations represent genus and specific epitaph, namely ALYDES (*Alyssum desertoides*), BROINE (*Bromus inermis*), BROTEC (*Bromus tectorum*), CARNUT (*Carduus nutans*), CERTES (*Ceratocephala testiculata*), CHEALB (*Chenopodium album*), CHEGLA (*Chenopodium glaucum*), CIRARV (*Cirsium arvense*), DESSOP (*Descurania sophia*), ELAANG (*Elaeagnus angustifolia*), EUPESU (*Euphorbia esula*), HALGLO (*Halogeton glomeratus*), LEPPER (*Lepidium perfoliatum*), MELOFF (*Melilotus officinale*), POLAVI (*Polygonum aviculare*), SALTRA (*Salsola tragus*), SISALT (*Sysimbrium altimissium*), TAROFF (*Taraxacum officinale*), THIINT (*Thinopyrum intermedium*), THLARV (*Thlaspi arvense*), TRADUB (*Tragopogon dubius*).

Our remote sensing approach was guided by the need to identify cover and productivity associated with historic treatments and the additional fact that these sites were distributed across the landscape with high variability in documentation as well as environmental conditions. Greenness indices such as the normalized difference vegetation index (NDVI) can be acquired by satellite over large areas at relatively coarse scales, however this approach may miss important details, such as the period of rapid green-up following snow-free days (which may only be detectable at finer spatial and temporal scales). This period of early, green-up can influence habitat use (for example, elk movement, sage-grouse activity, etc.), so it could be an important indicator of seasonal habitat condition on treated and untreated areas.

To enhance our resolution of this phenomenon, we are developing field-plot level, near-surface sensors to closely monitor changes in vegetation. In addition to detecting cheatgrass, this approach could provide important details of seasonal forage availability, for example, to determine when to stop elk feeding on state feed-grounds, where earlier feeding end dates are associated with reduced Brucellosis prevalence (Cross and others 2007). In addition, near-surface sensing platforms can target specific species (for example, perennial grasses or shrubs) or features (for example, bare soil, which is likely to show green-up by annuals including weeds), which remote sensing cannot, and specific species may be more or less palatable and thus more or less likely to provide forage/habitat for animal species of interest.

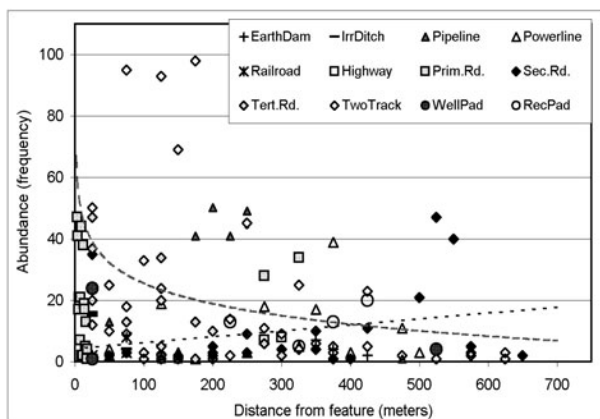


Figure 6. The distribution of cheatgrass (*Bromus tectorum*) relative to rural, land-use features in southwestern, Wyoming. The curve (dashed-line) demonstrates the negative, log-linear relationship between distance and abundance of cheatgrass.

Whereas, the straight line (dotted-line) clearly demonstrates, with a positive trend, that cheatgrass abundance did not decrease in abundance relative to all features. Variability in these distributions demonstrate the influence of other environmental factors. Sampled features (with abbreviation) include small, earthen dams (EarthDam), irrigation ditches (IrrDitch), oil and gas pipelines (Pipeline), overhead electrical lines (Powerline), Railroad, Highways, primary, paved thoroughfares (Prim.Rd.), large gravel roads (Sec.Rd.), small (short) gravel roads, driveways, spurs and access roads (Tert.Rd.), informal, unmaintained roads (Two-Track), active oil and gas facilities (WellPad) and reclaimed, former oil and gas facilities (RecPad).

As climate driven changes (for example, earlier snow melt) interact with vegetation, we expect plant phenology to shift in response to water availability and suitable growing conditions. This may make forage available earlier, for example, but it may also result in earlier senescence, or shifts in dominance to less-palatable, weedy species. To monitor these interactions, we established 50 multi-scale vegetation plots (Barnett and others 2007) in the vicinity of the Fall Creek feed-ground near Pinedale, Wyoming. These sites included burned and herbicide treated areas. We collected reflectance data from native and non-native vegetation using 14 mantis platforms (an adjustable tripod structure mounted with a multispectral camera to collect spectral reflectance data like a satellite from surface environments) during the 2010 growing season. We used “ground-truth” plot and reflectance data to measure correlations with remotely sensed data. We established an additional 30 plots in 5 historic treatment areas on and around the Pinedale Anticline to measure differences in plant species composition and cover as well as exposure of bare mineral soil.

Our remote sensing efforts were complimented by field research into composition and wildlife utilization. Since 1990, numerous restoration and enhancement projects have been implemented in the Little Mountain Ecosystem area (south of Rock Springs, Wyoming). Many of these projects involved prescribed burns to reduce sagebrush cover, increase herbaceous cover, increase other mountain shrub species (for example serviceberry, antelope bitterbrush), and retard the expansion of junipers into sagebrush. Wildfires and prescribed burns have been linked with the expansion of cheatgrass in similar systems in the Great Basin; however, in some situations burning has been

documented to support more stable plant communities that resist cheatgrass and other invasive plant species (Shinneman and Baker, 2009). We worked with land management agencies to map burn treatments in the Little Mountain area (approximately 25 miles south of Rock Springs, Wyoming) from 1990 through 2008. Using the design and sampling methods described in the previous section (Land Use and Invasive Plants), we sampled 22 vegetation transects (June through August) that were randomly distributed across burn treatments. We augmented the methodology described in the previous section at 17 of the 22 sites to include soil sampling (for determining soil texture and chemistry) and document the presence of biological soil crusts. Biological soil crusts, which can be disturbed through burning, are thought to help resist invasive species (Ponzetti and others 2007); therefore, a lack of crust may be associated with increased invasion potential. Sage-grouse pellet count surveys were conducted on two treatments, mowing and Tebuthiuron (herbicide, brand name "Spike"™), applied to sagebrush habitats in southwest Wyoming to ascertain use patterns and long-term trends associated with sage-grouse and treatment characteristics and gradients of energy development. Treatments were conducted on federal lands within the Moxa Arch Natural Gas Development near Granger, Wyoming. Treatment sites (implemented during 1997 through 2002) represented upland habitats dominated by Wyoming big sagebrush within areas selected by sage-grouse for nesting and early brood rearing. During 2009, forty-four 100-m by 4-m belt transects were randomly selected at mowed and Tebuthiuron applied treatment sites to evaluate sage-grouse use and the role of treated patch size, treated patch shape, and patch distance to lek (an assembly area for communal courtship display) or nesting habitat, and energy infrastructure.

RESULTS AND DISCUSSION

Preliminary results from the near-surface reflectance measurements indicated that we can track major phenological events such as flowering in addition to green-up and senescence using remote sensors. Vegetation plot sampling data representing one treated area (1960; figure 7) suggested that the sagebrush reduction treatment effects persist. Although not statistically significant the percent cover of Wyoming big sagebrush was lower in the treated (16 percent) than untreated area (27 percent), and

total vegetation cover followed the same pattern (33 percent cover treated vs. 54 percent cover untreated) and the difference was actually visible in remotely sensed imagery (figure 7). The percent cover of bare soil was significantly greater in the treated area (56 percent treated vs. 23 percent untreated; $p < 0.01$).

Preliminary results from assessments of burn treatments indicated a mixed response to cheatgrass invasion. Cheatgrass occurred in all transects but the frequency within subplots varied. Sixteen transects had a sandy loam texture while only one transect was classified as having a sandy clay loam. Subplot frequency of cheatgrass will be compared to duration since treatment and with soil nutrients and carbon/nitrogen ratios in future analyses

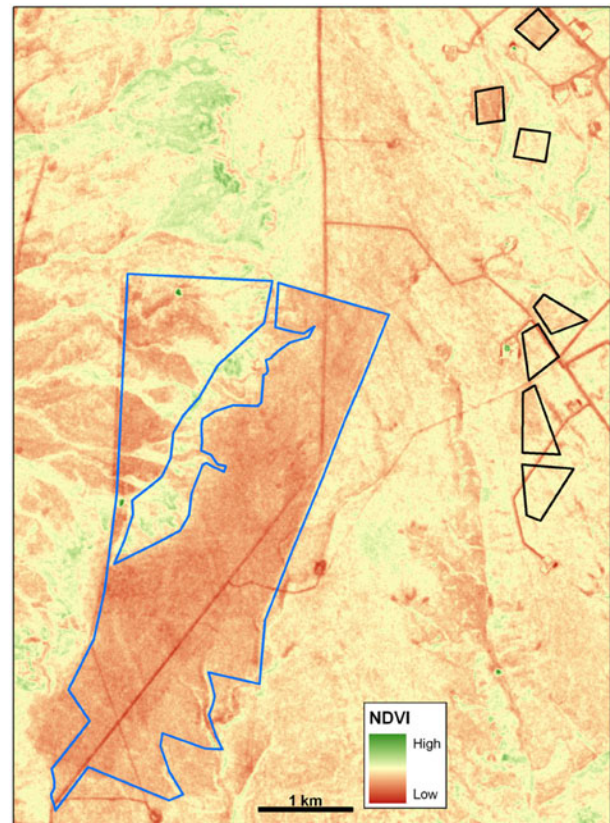


Figure 7. The normalized difference vegetation index (NDVI) is an indicator of greenness and standing vegetation. In this 2007 SPOT satellite image of an area that was sprayed with herbicide in 1960 (large, irregular black outlined area) the treated area is less green (index displayed as red) than the surrounding, untreated area. Note that roads and portions of well pads are also red (little or no vegetation). Image prepared by Mark Drummond, USGS.

Sage-grouse use surveys indicated that they are using mowed and Tebuthiuron treated areas and areas adjacent to energy infrastructure, however use appears to be connected to prior occupancy. Sage-grouse use surveys also indicated that they are using large open areas in the center of treatments less frequently than the edge of treatments near the cover provided by untreated sagebrush. Treated sites were most frequently used by sage-grouse during nesting and brood rearing with limited use during fall and winter. Future analyses will include the expansion of additional treatment areas (sampling conducted during 2010) to evaluate if differences exist between treatment types, season of use, proximity to leks and prolonged effects of energy infrastructure.

CONCLUSIONS

While there is a lot of sagebrush on the map, much of it is fragmented, manipulated and impacted by biological invasions induced by perpetual and widespread surface disturbances. The extent of the "sagebrush sea" was greatly reduced in extent before this research began (Connelly and others 2004), making understanding and effective management of these lands important for wildlife conservation. By using a combination of field sampling and remote sensor platforms, we developed detailed cover estimations for shrub habitat components across large regions (State of Wyoming) that accurately depict the current distribution of sagebrush and associated habitats. These data greatly improved the resolution, accuracy and information content of existing products, exhibiting detailed projections within 10 percent of actual cover in most locations. Continuous cover projections, as compared to type-mapping, provide a comprehensive perspective of the heterogeneous distribution of vegetation, litter and bare ground within sagebrush communities, identifying areas of both high and low cover. In the future, it is hoped that these methods, and data, will form a baseline for monitoring changes on this landscape. The U.S. Geological Survey has initiated research applying our component estimates to current and historical vegetation change, climate variation, sage grouse habitat distribution, and grazing trends.

The potential distribution of weeds, especially annual grasses, across the sagebrush steppe is widespread with intense local infestations. Our data showed increased abundance of noxious, invasive plants adjacent to anthropogenic features, especially roads

and well pads. However, surveying a wide region and variety of invaders brings recognition that there is not a single profile for invasive plants, even across a consolidated, semi-arid region. For the most abundant, problematic species such as cheatgrass, desert alysium and halogeton, it was not clear that the distribution of infrastructure remains an important driver of distributions because these species were often observed to be abundant hundreds of meters away from the nearest feature. While the distance effect was significant for most of these cases (linear and log-linear), residual variability in these models indicated that other, widespread environmental conditions or activities were also responsible for patterns of invasive plant distribution in this region. Treated sites were most frequently used by sage-grouse during nesting and brood rearing with limited use during fall and winter, and surveys also indicated that they used large open areas less frequently than the edge of treatments. Results also indicated a mixed response of treatments to cheatgrass invasion, however weeds were observed on every treated site. Clearly there are potentially important interactions between habitat distributions, habitat treatments, invasive plants and use of habitats by wildlife. These studies begin to elucidate these patterns and their interactions.

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First Year Soil Impacts of Well-Pad Development and Reclamation on Wyoming's Sagebrush Steppe

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ABSTRACT

In recent years, natural gas extraction activities have disturbed thousands of acres of arid and semiarid regions in Wyoming's sagebrush steppe ecosystem. Thin, nutrient poor topsoils, combined with subsoils potentially high in salts, limit the resilience of these arid and semiarid soil systems. Stripping, stockpiling, and respreading topsoil stimulates decomposition and loss of soil organic matter (SOM) by breaking apart soil structure and eliminating inputs of plant residues, which can result in reduced SOM content. When the soil structure is disturbed organic matter can rapidly decompose, releasing mineral nutrients that are mobile and can be lost to weeds, leaching, erosion, or volatilization. The purpose of this study is to gain an understanding of how natural gas development and reclamation activities impact soil properties, plant growth re-establishment, and the ability of disturbed sagebrush ecosystems to recover over time. Soil samples were collected from stockpiles, respread topsoil and adjacent undisturbed areas from three natural gas fields located in Western Wyoming. Results suggest that soil organic matter needed for plant growth becomes mineralized or released when the soil is disturbed. The data show a small increase in plant-available mineral nitrogen (N) concentrations after stripping and stockpiling compared to undisturbed soils, and then a large increase in available N following respreading for reclamation. This suggests that easily decomposable organic matter exposed by destruction of soil structure during stripping is conserved in deep stockpiles but then rapidly decomposed upon re-exposure to air and moisture with respreading. The spike in mineral N likely originates from organic compounds that, in undisturbed conditions, hold and slowly release N and other nutrients. It represents a significant potential loss of this important "time-release" nutrient pool. The spike in mineral nutrients probably stimulates prolific weed production often observed on reclaimed sites. Weeds that stay and decompose on site may conserve and recycle the nutrients, but the data suggest a need for a better way to accomplish this.

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INTRODUCTION

Much ecological disturbance in the western U. S. is related to natural gas production, coal mining, or other energy development and is located in arid and/or semi-arid regions. These ecoregions that occupy much of the western landscape are difficult to reclaim once they are disturbed (Bunting and others 2003; Whisenant 1999). Low soil fertility and organic matter contents, slow-growing and difficult-to-establish vegetation, saline or sodic conditions, and other constraints related to low rainfall create fragile conditions, with low resistance to and resilience after disruption. In recent years, natural gas extraction activities have disturbed thousands of acres of arid and semiarid regions in Wyoming's sagebrush steppe ecosystem. The extraction of natural gas is a short but drastic perturbation to soil processes and the terrestrial ecosystem. In addition because of the

infrastructure associated with wells (i.e. well pads, roads, and pipelines) energy development potentially influences ecoregions indirectly by exotic plant establishment or directly by the loss of wintering and breeding habitat for wildlife as well as migration barriers for ungulates (Berger 2003, 2004; Lyon and Anderson 2003). During natural gas well pad development, topsoil, which provides the majority of nutrients essential for plant growth, is typically stripped, stockpiled, and respread for reclamation. Vegetation and topsoil are removed using heavy operating equipment and stockpiled on the well pad until drilling is complete and then respread and seeded for reclamation. We speculate that stripping, stockpiling, and respreading of topsoil disrupts soil structure that protects labile organic carbon (C) and N. Labile organic C and N are protected from degradation within soil aggregates, but become mineralized when disturbed which may result in a shift

in the C and N dynamics that exist in SOM pools. Soil organic matter is an important nutrient pool that plays a critical role in ecosystem stability, including nutrient cycling, soil structure formation, soil water holding capacity, energy for microorganisms, and essential nutrients required for plant growth.

There has been much research conducted on energy related disturbance impacts to soil, however, this study investigates the immediate (<1 yr) effects that occur on the redistribution of the SOM pools during the different phases (stripping, stockpiling, and resspreading) of well pad development. Understanding how disturbance alters SOM pools will contribute to greater reclamation success and ecosystem recovery. The objectives of this study are to 1) determine effects of stockpiling depths on C and N dynamics and 2) quantify effects of stripping, stockpiling, and resspreading on soil C and N dynamics.

MATERIALS AND METHODS

Site Information and Field Sampling

Study Area

Nine well pads were selected from three Wyoming natural gas fields: Pinedale Anticline (Anticline), Jonah, and Wamsutter. Each site location consisted of three stockpiles (SP), three recently reclaimed well pads (RC), and three adjacent undisturbed sites (UN). Soil samples were collected from stockpiles, resspread topsoil and adjacent undisturbed sites in 2009 and 2010. For ecological site descriptions and climate data for each site location refer to Driessen and others (this volume).

Stockpile Sampling

Stockpiled topsoil soil samples were collected from <1 yr (Jonah and Wamsutter) and <5 yr (Anticline) old stockpiles. On each stockpile, three randomly located holes were augured to a depth of 250 cm. Samples were bulked by depths of 0-5 cm, 5-20 cm, 20-100 cm, 100-200 cm, and 200-250 cm for each of the nine stockpiles. An adjacent undisturbed site was also randomly sampled with an auger to serve as a reference soil. From the undisturbed site, a composite soil sample was collected from 0-20 cm to represent the topsoil stripping depth.

Reclaimed Well Pad Sampling

After the stockpiled topsoil was resspread and seeded for reclamation in Fall 2009, soil samples were collected on the recently resspread topsoil and adjacent undisturbed area along three transects set up on a 0.1 ha plot. Soil samples were collected from 0-5 cm, 5-20 cm, and 20-30 cm at three points along each of three 32 m transects. Soil samples were bulked by depth for each transect, thus a total of 9 samples were collected from each plot.

Laboratory Analyses

Soil samples were kept at 4°C until they were brought back to the lab for analysis. Ten grams of field moist soil was measured for gravimetric moisture content (Gardner 1986) and mineral N. Mineral N, an index of plant-available N, was extracted from 10-g subsamples with 50 mL of K₂SO₄ and run on a microplate spectrophotometer (Powerwave HT, BioTek Instruments, Winooski, Vermont) for NH₄-N (Weatherburn 1967) and NO₃-N (Doan and Horwath, 2003). An additional 22 g of field moist soil was measured for labile organic C and N determination using aerobic incubation (Hart and others 1994; Zibilske 1994). Samples were brought to 23 percent gravimetric moisture content prior to incubation. Aerobic incubations yielded mineralizable N and C after 14 d under optimal water and temperature conditions. Samples were incubated in sealed jars and jar lids were fitted with rubber septa for the collection of gas samples. Headspace samples (30 ml) were collected in syringes fitted with gas-tight valves after mixing the total volume by plunging the syringe up and down. Samples were collected on day 1, 4, 7, and 14 to measure potentially mineralizable C or labile organic C. All incubation jars were flushed and refilled with ambient air following each sampling. Four blank jars (no soil) were included in each experiment to control for background CO₂ concentration. Headspace samples were analyzed for CO₂ concentration using an infrared gas analyzer and calibrated with three standard gases (Model LI-820, LICOR Inc., Lincoln, Nebraska). After the 14-d incubation period, a 10-g subsample was taken from the 22-g sample to determine gravimetric moisture content after 14 d. The remaining soil was extracted with 50 mL of K₂SO₄ and analyzed for NH₄-N and NO₃-N as described for mineral N above. This represents the amount of organic N mineralized under optimal conditions after a 14-d incubation period. Potentially mineralizable N or labile organic N is

achieved by subtracting the initial inorganic N content from the N content after the 14-d incubation period.

Statistical Analysis

The data were analyzed statistically using one way analysis of variance using SAS 9.1.3 SP4 (SAS Institute 2008). All statistical tests were conducted at $P < 0.05$.

RESULTS AND DISCUSSION

Stockpile Depth Effects

The stockpile data presented reflects the average midpoint of each depth interval compiled for all three site locations. Although not significant, mineral N increased slightly with increasing stockpile depth for the Jonah and Wamsutter sites (figure 1a). Mineral N

for the Anticline increased with increasing depth, but declined beyond 150-cm depth. Abdul-Kareem and McRae (1984) reported that $\text{NO}_3\text{-N}$ concentrations in stockpiles were similar to those in adjacent undisturbed soils, but $\text{NH}_4\text{-N}$ was greater with depth in all stockpiles when compared to adjacent undisturbed soils.

Labile organic C and N concentrations increased with increasing stockpiling depth (figure 1b and 1c), suggesting that the labile SOM pool is protected and being conserved deep in stockpiles. Other research (Abdul-Kareem and McRae 1984; Visser and others 1984; Williamson and Johnson 1990) has shown greater soil respiration rates deeper in stockpiles than at the surface of stockpiles. Management implications often recommend shallow topsoil stockpiles, but our data suggests that may not be necessary.

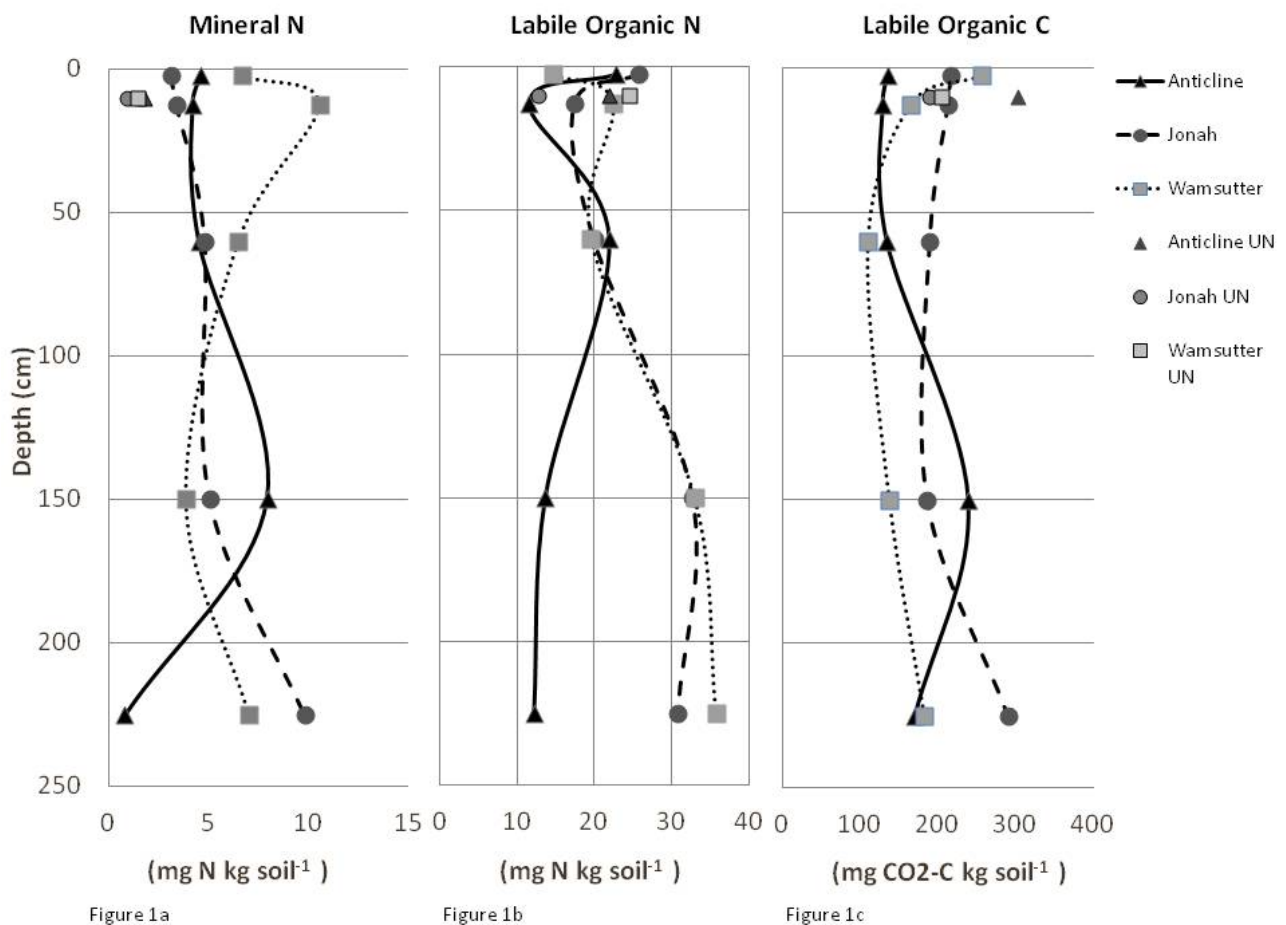


Figure 1. Average mineral N(a), labile organic N (b) (mg N kg soil⁻¹) and labile organic C (c) concentrations (cumulative mg CO₂-C kg soil⁻¹ during 14-d incubation) from stockpile depths and adjacent undisturbed from each natural gas field.

Stripping, Stockpiling, and Respreading Effects

For all three site locations mineral N was significantly (Anticline $p = 0.0052$, Jonah $p = 0.0106$, Wamsutter $p = 0.0018$) greater for the reclaimed treatment than in the undisturbed and stockpile treatments. The data shows a small increase in mineral N concentrations after stripping and stockpiling compared to undisturbed soils, and then a large increase in available N following re-spreading for reclamation (figure 2a). Soil organic matter is decomposed upon re-exposure to air and moisture with respreading. Williamson and Johnson (1990) reported that decomposition occurred as a result of the labile organic matter and mineral N release associated with stockpile disturbance and restoration. In addition, the $\text{NH}_4\text{-N}$ that accumulated within the stockpile was converted to $\text{NO}_3\text{-N}$ as oxygen became available during the restoration process (Williamson and Johnson 1990).

Labile organic N concentrations were significantly ($p = 0.0076$) less for the stockpile and reclaimed treatments than the undisturbed reference site at the Anticline (figure 2b). Labile organic N concentrations significantly increased in the stockpile treatment compared to the reclaimed treatment for the Wamsutter ($p = 0.0145$) and Jonah ($p = 0.0341$) gas fields. In two of the three sites (Jonah and Wamsutter) labile organic N concentrations were greater in the stockpiles than in the undisturbed reference sites. Ingram and others (2005) found lower labile organic N in stockpiles than in native sites. Furthermore, Lindemann and others (1989) showed slightly lower labile organic N concentrations in stockpiled topsoil compared to fresh topsoil.

All three site locations had significantly (Jonah $p = 0.0120$, Anticline $p = 0.0366$, Wamsutter $p = 0.0379$) lower labile organic C concentrations in the reclaimed plots than in the undisturbed plots (figure 2c). Our data show labile organic C concentrations were less in the stockpiles than the undisturbed sites. Ingram and others (2005) reported carbon mineralization rates were greater in a 2-yr-old stockpile than native sites after 21 days of incubation. Differences in labile organic C and N concentrations between the native and reclaimed sites may be due to differences in microbial communities, break-up of microaggregates, or the addition of new non-humified plant residues (Ingram and others 2005). Two of the three sites

(Jonah and Wamsutter) showed that loss of labile organic C and N was most pronounced upon re-spreading stockpiled soil, whereas on the Anticline this loss occurred upon stockpiling. This loss probably reflects the fact that stockpiles on the Anticline were older and had been moved several times causing redisturbance and loss of labile organic C and N.

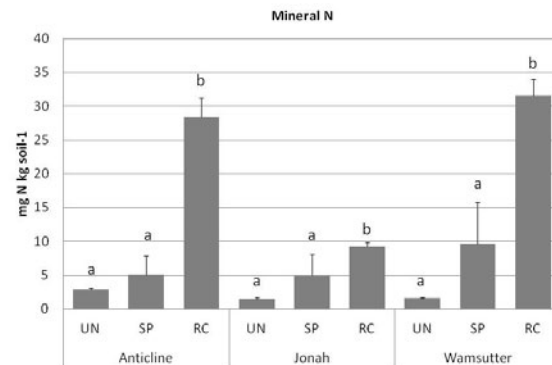


Figure 2a

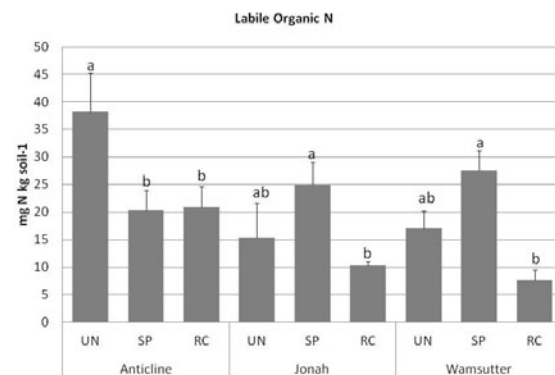


Figure 2b

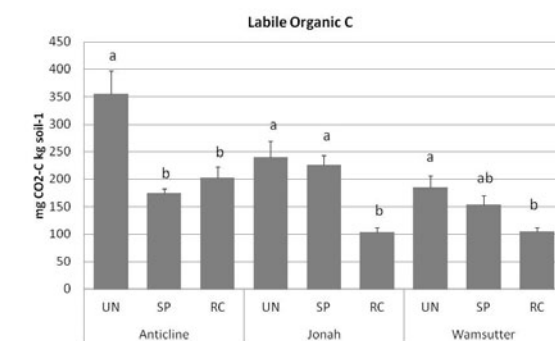


Figure 2c

Figure 2. Average Mineral N (a), labile organic N (b) (mg N kg soil^{-1}) and labile organic C (c) ($\text{mg CO}_2\text{-C kg soil}^{-1}$) during 14-d incubation) from undisturbed, stockpile, and reclaimed plots from each natural gas field. Letters indicate significance differences ($P \leq 0.05$) between undisturbed, stockpiles, and reclaimed plots at each gas field. Error bars denote standard error.

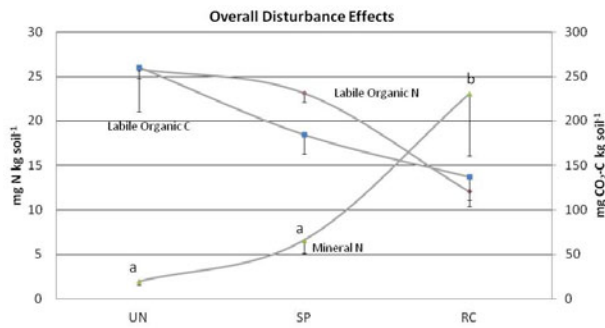


Figure 3. Average mineral N, labile organic N (mg N kg soil^{-1}) and labile organic C concentrations (cumulative $\text{mg CO}_2\text{-C kg soil}^{-1}$ during 14-d incubation) from undisturbed, stockpile, and reclaimed plots for all site locations. Letters indicate significance differences ($P \leq 0.05$) between undisturbed, stockpiles, and reclaimed plots at all 3 gas fields. Error bars denote standard error.

Overall Disturbance Effects

The labile pool of SOM is a reservoir of time-release nutrients and is extremely important for ecosystem resiliency. The data presented in Figure 3 represents the mineral N and labile organic C and N concentrations averaged and compiled for all 3 site locations. The data show that mineral N increases or becomes available with each phase of disturbance. Mineral N is significantly greater ($p = 0.0254$) for the reclaimed treatment than the undisturbed and stockpile treatments. Although not significant, the data show that labile organic C and N are reduced with each phase of disturbance. The active pool consists of readily available nutrients whereas the slow pool is less available for microbial degradation because it is protected in the micro- and macro-aggregates. Thus breaking soil aggregates releases a labile organic material (Beare and others 1994; Kristensen and others 2000), changing nutrient pools (Chapin and others 2002).

CONCLUSION

The SOM that is needed for plant growth becomes mineralized and released when the soil is disturbed. The destruction of soil aggregates stimulates mineralization and decomposition resulting in reduced C and N (Chapin and others 2002; Ingram and others 2005; Wick and others 2009) and SOM (Abdul-

Kareem and McRae 1984). Our data indicate that the initial stripping of topsoil disrupts the soil structure causing an increase in the labile organic C and N when compared to the undisturbed reference site. Once the topsoil is stockpiled the labile organic C and N is protected from mineralization deep within the stockpile. However, labile organic C and N concentrations are reduced when stockpiled and re-spread for reclamation, suggesting that the protected pool is being mineralized and lost to the environment. Losses in labile organic C and N are greatest just beneath the surface where moisture, temperature, and aeration are probably optimal for mineralization during the time soil is stockpiled. Mineralization increases with each disturbance activity, but is greater when the topsoil is respread and tilled for seeding. The spike in mineral N originates from organic compounds that, in undisturbed conditions, hold and slowly release N and other nutrients. The spike in mineral nutrients probably stimulates prolific weed production often observed on reclaimed sites.

The data suggest there is a loss of valuable SOM in soils and an untimely release of nutrients. The data indicates a need for alternative handling and/or management methods that conserve labile SOM and mineral nutrients, such as less destructive stripping/spreading methods that conserve soil structure, and cover crops or C additions that immobilize mineral N and keep it on site. Stahl and others (2002) stated that successful restoration of a disturbed area is dependent on maintenance of soil quality and minimizing the human footprint to soil resources could prevent further site degradation and facilitate site restoration.

ACKNOWLEDGEMENTS

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Immediate Effects of Controlled Livestock Treatment on Reclaimed Natural Gas Well Pads

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ABSTRACT

Wyoming shrublands have undergone extensive energy development in recent years. Much of this development occurs on public land designated for multiple uses. Reclamation of these areas has proven difficult due to the harsh climate and alteration of the thin, nutrient poor topsoil during development activities. Energy development and reclamation activities often lead to topsoil dilution, rapid mineralization of nutrients and soil organic matter (SOM), and loss of soil structure. These changes have the potential to degrade the suitability of the soil as a medium to sustain a desirable plant community. Reclamation of land disturbed for energy development in this area has largely been executed by the extraction companies and evaluated by the governing agency (typically, the Bureau of Land Management (BLM)). Other parties who rely on this land, such as ranchers with grazing permits, are not typically involved in reclamation. In this study, we examine an unconventional reclamation technique that aims to involve ranchers in the reclamation process: controlled livestock impact. The theory behind this technique is that by confining livestock on a seeded and reclaimed site the animals will improve the seedbed and seed to soil contact through fertilization and hoof action. Natural gas well pads that were reclaimed in the fall of 2009 were selected from three Wyoming natural gas fields. Two treatment plots were established on each well pad: traditionally reclaimed and reclaimed with the cattle impact treatment. Cattle treatments were applied in fall 2009 immediately after reclamation and seeding. Soil samples were taken from the reclaimed plots and before and after the cattle impact on treated plots. Soil samples were then analyzed for SOM parameters including percent light fraction organic matter (LF) and labile C and N. Post-cattle treatment plots had more mineralizable C and more N variability than pre-cattle plots, which indicates an impact from the cattle treatment on SOM characteristics.

In Monaco, T.A. et al. comps. 2011. Proceedings – Threats to Shrubland Ecosystem Integrity; 2010 May 18-20; Logan, UT. Natural Resources and Environmental Issues, Volume XVII. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan Utah, USA.

INTRODUCTION

Natural Gas Well Pad Reclamation in Wyoming

Wyoming is one of the nations' leaders in natural gas production and proven reserves. The Energy Information Administration reports that Wyoming ranks second in the United States for proven dry natural gas reserves as of 2008 (USEIA 2009). Natural gas development in Wyoming occurs on state, federal, and private land, which is often used for livestock grazing, wildlife habitat, recreation, and other activities in addition to resource extraction. Natural gas extraction in Southwest Wyoming requires a level area for drilling activities (well pad), pipelines for transport of resources, and access roads for maintenance. The nature of this type of land

disturbance has resulted in habitat loss or fragmentation (Walston and others 2009), wildlife avoidance (Lyon and Anderson 2003; Sawyer and others 2009), changes in plant communities (Bergquist and others 2007), and other indirect consequences. Thus, techniques that accelerate the successful reclamation of these sites are highly desirable.

Reclamation of Wyoming's shrublands is often difficult because of harsh climate, nutrient poor topsoil, changes in soil properties during development and reclamation activities, herbivory, and lack of viable seed. Many mechanisms have been explored to ameliorate these issues, but are often expensive or difficult to implement. Furthermore, other affected parties, such as ranchers, are rarely incorporated into reclamation plans although their livelihoods may depend on successful reclamation.

Controlled Livestock Impact

Controlled livestock impact has gained popularity with land managers in recent years as a reclamation tool. A large collection of testimonial evidence exists proclaiming the success of using animals to prepare the seedbed and maintain a desirable plant community, but little science-based research has assessed these claims. Controlled livestock impact is different from grazing, as it is a treatment applied to a site with little to no standing forage. Grazing or browsing animals are confined at high density on a reclaimed area and are fed, and often allowed to bed down, on the location. The idea behind this is that the combination of hoof action and addition of organic materials will improve soil conditions for plant establishment. Seeding may occur before or after the livestock impact treatment, or mature native grass hay may be used to both feed the livestock and provide seed to the area.

This study aims to quantify the immediate effects of a controlled livestock impact on basic soil organic matter (SOM) characteristics. SOM is important for plant establishment on reclaimed locations as it provides nutrients, improves water holding capacity of soils, and reduces erosion by promoting aggregation of soil particles. Moreover, reclaimed soils in Wyoming have been shown to have lower SOM than comparable undisturbed soils (Anderson and others 2008; Ingram and others 2005; Mummey and Stahl 2004; Stahl and others 2002; Wick and others 2009a; Wick and others 2009b). This study was designed to assess the immediate effects of controlled livestock impact on SOM, thus we focused our efforts on characterizing the labile organic matter pools. Labile and light fraction (LF) organic matter pools reflect changes in topsoil management and are good indicators of topsoil quality (Sohi and others 2010). We expect that both the labile organic C and N pools

will be higher after the cattle treatment than before. We believe that contributions of waste feed and excrement will increase the amount of C and N in the labile organic pool. Furthermore, we hypothesize that there will be more LF after livestock treatment for similar reasons.

METHODS

Study Area

Ten well pads were selected from three Wyoming natural gas fields: Pinedale Anticline (Anticline), Jonah, and Wamsutter. The pre-disturbance ecological site descriptions for the Anticline well pads are loamy or shallow loamy 10 to 14-inch Foothills and Basins and clayey or gravelly 7 to 9-inch Green River and Great Divide; and either clayey or loamy 7 to 9-inch Green River and Great Divide for the Jonah. (NRCS 2009). The NRCS has not yet classified the ecological sites for the Wamsutter area, but we found the soil to have sandy loam texture and the dominant vegetation is Wyoming big sagebrush (*Artemisia tridentata* spp. *wyomingensis*) or Gardener's saltbush (*Atriplex gardnerii*). All of the fields are cool and dry with the majority of the annual precipitation occurring as snowfall (table 1).

Sampling Design

Each well pad was assigned two treatment plots on the reclaimed area, one of which received the controlled livestock impact treatment (cattle) and one which did not (reclaimed). The cattle plots were sampled before (pre-cattle) and after (post-cattle) the livestock treatment was implemented. Plots were 0.10 ha (0.25 ac) in size with three, 34 m (112 ft) permanent transects. Soil samples were taken at 0 to 5-cm (0 to 2.5-inch) depth at three locations along transects and bulked by transect.

Table 1. Climate information for the three natural gas fields. Data obtained from Western Regional Climate Center from nearest data loggers to each gas field based on averages from 1948 to 2005.

Site	Average Max Temp °C (°F)	Average Min Temp °C (°F)	Mean Annual Precip mm (in)
Anticline	10.9 (51.7)	-6.72 (19.9)	277 (10.9)
Jonah	12.7 (54.8)	-6.50 (20.3)	187 (7.35)
Wamsutter	12.9 (55.3)	-2.61 (27.3)	174 (6.84)

Controlled Livestock Impact Treatment

The well pads were reclaimed and seeded in the fall of 2009. Topsoil handling and storage, seed mixes and seeding rates, and mulching varied between natural gas fields due to differences in company policies and governing legislation. The cattle treatment was superimposed on the traditional reclamation and seeding for each field. Cattle plots were temporarily fenced and certified weed-free hay was scattered throughout the fenced area. On the Jonah and Anticline production areas, 25 cows occupied the 0.10 ha (0.25 ac) plots for 24 hrs; while 12 bulls occupied the Wamsutter area plots for 48 hrs. This stocking rate was determined by estimating the amount of organic matter lost through construction and reclamation activities and then calculating how much organic material; in the form of feces, urine, and excess feed; a single cow contributes in a day. According to the Natural Resources Conservation Service (NRCS), a typical 453.6 kg (1000 lb) beef cow produces 4.85 kg (10.7 lb) of manure per day (NRCS 2010), which yields 1.13 kg (2.5 lbs) of dry organic material per 453.6 kg animal per day (van Vliet and others 2007). Also, cattle typically waste about 30 percent of total hay fed on the ground, or as much as 8.16 kg (18 lbs) per animal per day for low-quality forage. Data from reclaimed coalmines suggest that 35 to 69 percent of SOM is lost by the time the soil is reclaimed (Anderson and others 2008; Ingram and others 2005; Mummey and Stahl 2004; Wick and others 2009a; Wick and others 2009b). Assuming a bulk density of 1.3 g cm^{-3} and an initial SOM content of 1.5 percent, 183 to 362 cattle $\text{ha}^{-1} \text{ d}^{-1}$ (74 to 147 cattle $\text{ac}^{-1} \text{ d}^{-1}$) would be required to replace the organic matter loss. We adjusted our final stocking rate of 240 cattle $\text{ha}^{-1} \text{ d}^{-1}$ (100 cattle $\text{ac}^{-1} \text{ d}^{-1}$) after discussing feasible rates with the cattle producers who cooperated with this project.

Laboratory Analyses

Soil samples were immediately chilled at 4 °C upon collection until they reached the laboratory. Soil was then divided for field-moist analyses and dry analyses. Field moist samples were used for assessing labile organic pools. Dry samples were dried at room temperature for 48 h, and then sieved to 6.35 mm (0.25 inch) for LF analysis.

Labile Organic C and N

Approximately 10 g of field moist samples were immediately extracted with 50 mL of K_2SO_4 using Q5 filters, upon returning to the lab. This analysis allows quantification of bio-available N that is immediately available in the soil, which is also known as mineral N. Another 10 g was used to determine gravimetric moisture content (Gardner 1986) so samples could be normalized for moisture content. Extracts were frozen for storage and then run on a microplate spectrophotometer (Powerwave HT, BioTek Instruments, Winooski, Vermont) for $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$ as described by Larios (2008). For NH_4 , 40 mL of sample was mixed with 80 mL of sodium salicylate solution and 80 mL of bleach- NaOH solution and allowed to develop color for 1 hr before reading on the spectrophotometer. NO_3 analysis used 10 mL of sample to 190 mL of $\text{VCl}_3\text{-HCl}$ solution (Doane and Horwath 2003) and was developed for 18 hr before reading.

Twenty-two g of field moist soil was brought to approximately 23-percent moisture content for labile C and N. Soil underwent a 14-day aerobic incubation as described in Zibilske (1994) and Hart and others (1994). Carbon dioxide samples were drawn out using 30-ml syringes through the rubber septa in the incubation jars on the first, fourth, seventh, eleventh, and last days of the incubation period. These samples were then analyzed on an infrared gas CO_2 analyzer (LI-820, LI-COR Inc, Lincoln, Nebraska) on the days they were withdrawn. A 10-g sub-sample of the 22 g sample was analyzed for gravimetric moisture at the end of the 14-day incubation period to correct for actual moisture content. The cumulative C released over the 14-day incubation period is the potentially mineralizable C, or labile organic C content of the soil.

After the 14-day incubation period, the remaining soil was extracted with 50 mL of K_2SO_4 . The sample was analyzed for NH_4 and NO_3 as described for mineral N above. This represents the amount of organic N mineralized under optimal conditions after a 14-day incubation period. Potentially mineralizable N or labile organic N is achieved by subtracting the initial inorganic N content from the N content after the 14-day incubation period.

LF

A 10-g sample from the dried and sieved soil was used for organic fraction analysis. The density fractionation method described by Sohi and others (2001), using 1.8 g cm^{-1} NaI, was used to obtain LF. Free LF (fLF) was collected from the surface of the solution after gentle mixing, while occluded LF (oLF) was collected after vigorous shaking and 110 seconds in a sonicator. Both forms of LF were centrifuged at 2000 rpm until mineral components of the sample settled to the bottom of the tube. Lids and rims of tubes were rinsed with more NaI and LF was collected using an aspirator. Samples were collected on a nylon 20-mm filter and rinsed thoroughly with deionized water. Samples were then dried in aluminum tins at 60°C (140°F), and weighed to 0.0001 g. These two fractions determined by density represent total LF (von Lutzow and others 2007).

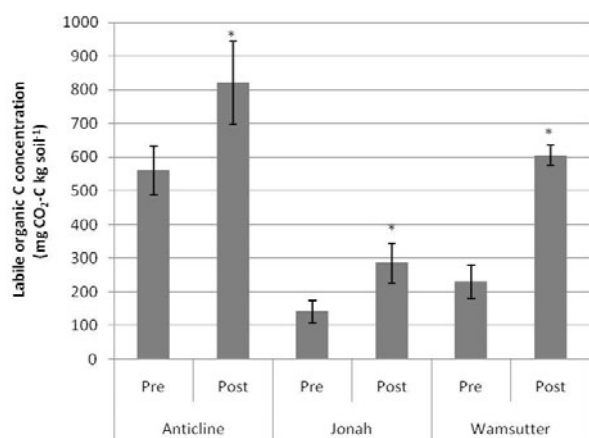


Figure 1. Labile organic carbon (cumulative mg CO₂-C kg soil⁻¹ during 14-d incubation) from pre-cattle and post-cattle sampling for the three natural gas fields. Error bars represent standard error. *significantly ($p < 0.05$) higher labile organic carbon between treatments within a natural gas field.

RESULTS

Paired t-tests were used to determine differences between the pre and post-cattle treatments. Statistical tests were based on treatment means and an alpha of 0.05 was used to determine significance.

Labile Organic C and N

Labile organic C concentrations were significantly higher after the cattle treatment for the Anticline ($p = 0.027$), Jonah ($p = 0.006$), and Wamsutter ($p = 0.010$). There were also noticeable differences in

labile organic C between the sites, with the highest on the Anticline, followed by Wamsutter, and finally, the Jonah (figure 1).

While differences between natural gas fields existed, there was no difference in mineral N after the cattle treatment within a location (figure 2a). Labile organic N was greater before the cattle treatment for the Jonah ($p = 0.003$) and no trends are observed between natural gas fields. The data for both the Anticline and Wamsutter do, however, suggest that there is increased variability in labile organic N after the cattle treatment (figure 2b).

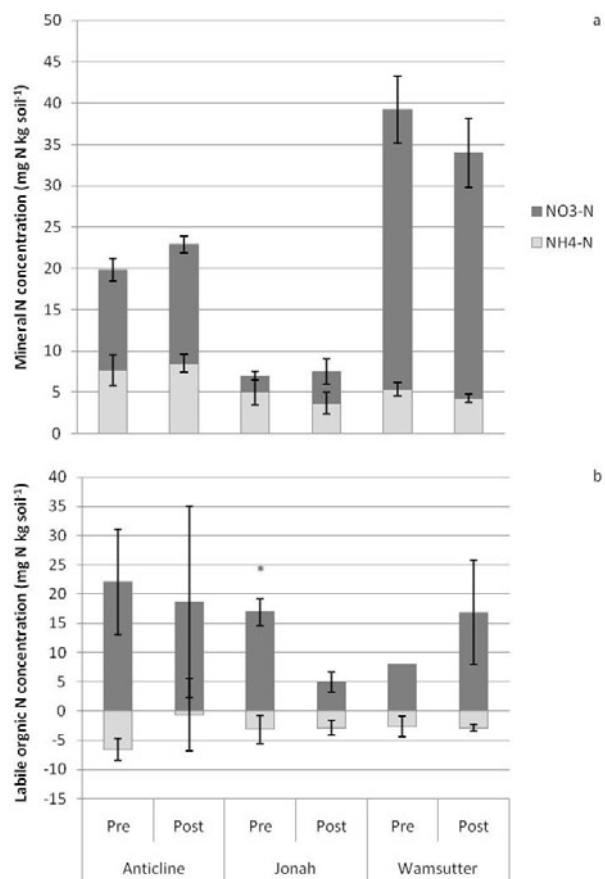


Figure 2. Mineral (a) and labile organic (b) N concentration (mg N kg soil⁻¹) in the forms of NO₃⁻ and NH₄⁺ for three natural gas fields before and after cattle treatment. Mineral N is the initial concentration of available N while labile organic N is the initial mineral N subtracted from mineral N after a 14 d aerobic incubation period. Negative NH₄⁺ values suggest nitrification occurred during the incubation period. Error bars denote standard error. *significantly ($p < 0.05$) higher N concentration between treatments within a natural gas field.

LF

There was no difference in fLF after the cattle treatment on any of the gas fields. Differences between gas fields are similar to those seen in the labile organic pool, with Wamsutter the highest, then the Anticline, and Jonah the lowest (figure 3a).

There was significantly higher oLF after the cattle treatment on the Jonah ($p = 0.006$), but not on the other two fields. Differences between fields follow the same trend seen in the other organic matter characteristics (figure 3b).

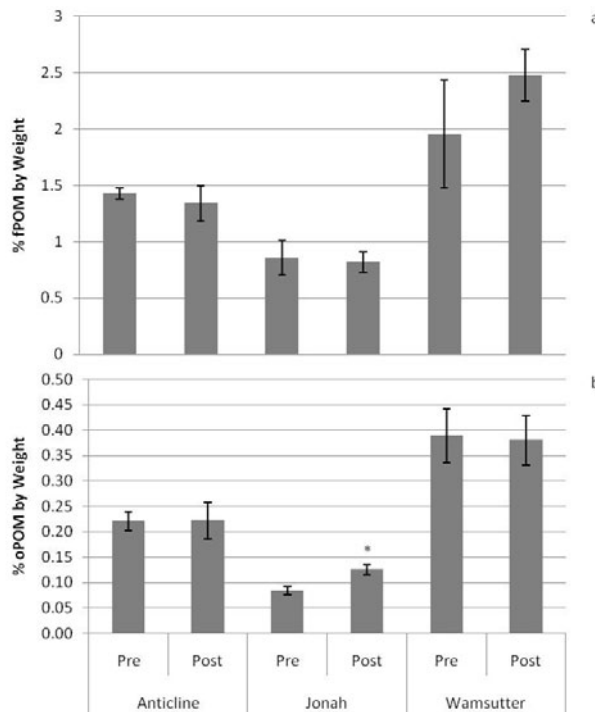


Figure 3. Free (a) and occluded (b) light fraction organic matter percent by weight before and after cattle treatment for three natural gas fields. Error bars indicate standard error. * significantly ($p < 0.05$) higher percent light fraction organic matter after treatment within a natural gas field.

DISCUSSION

Labile Organic C and N

As hypothesized, labile organic C content was higher post-cattle than pre-cattle. These results are similar to those of agricultural plots in a shrub-steppe ecosystem treated with composted dairy waste (Cochran and others 2007). In this agricultural study,

a 175-day incubation period revealed cultivated soils treated with dairy compost mineralized more C than untreated plots with native vegetation (Cochran and others 2007). Another agricultural study found that plots treated with additions of sewage sludge compost, dairy manure compost, and corn silage compost had higher total C and available C than untreated or conventionally fertilized plots (Lynch and others 2005).

Contrary to labile organic C pools, N pools did not agree with the hypothesis that N would be higher after the cattle treatment. This relationship yields a higher C: N in the labile pool, which is more similar to what is expected on native rangeland. One possible explanation for this is that N could have been immobilized or volatilized immediately after the treatment was applied. Burgos and others (2006) found this phenomenon to be true in sandy soils for two organic amendments. They observed that municipal sewage compost and agro-forest compost both initially immobilized N and then continuously released mineral N for the duration of the study (Burgos and others 2006). Continued sampling may reveal whether the cattle treatment amendment behaves similarly to other soil organic amendments. Labile organic N on the Anticline and Wamsutter fields may lack differences between treatments due to the high variability on the post-cattle plots. On the other hand, this variability may imply that the cattle treatment promotes heterogeneous soil conditions; which could be important for reinstating the patchiness of soil quality that naturally occurs on Wyoming shrublands (Burke 1989; Eviner and Hawkes 2008).

LF

No differences were observed between the pre and post-cattle data for fLF or oLF. The one exception was the occluded fraction on the Jonah where, as hypothesized, the oLF fraction was higher after the cattle treatment. Wick and others (2009a) found the highest amount of microaggregates, 53 to 250 μ m (0.002 to 0.010 inch), during the first year of reclamation. In spite of this fact, the first year after reclamation had the lowest amount of interaggregate LF C (Wick and others 2009a). The Jonah site could have more oLF after the cattle treatment because the organic additions may have been trapped during the formation of these first-year aggregates. Furthermore, the Jonah post-cattle treatment was the only location

to have significantly less labile organic N than before the cattle treatment. This may suggest that some of the labile N was not only immobilized by microbes, but also fixed in soil aggregates. Additional analyses on C and N content of the fLF and oLF fractions would provide more insight to the processes occurring on the treated plots.

CONCLUSIONS

In conclusion, the controlled livestock impact explored in this study had immediate effects on soil labile C and N and on LF pools. Whether or not these effects translate to achieving short-term reclamation goals remains to be seen. Soil and vegetation parameters will be continually monitored during the 2010 growing season. Results from these and further analyses of the 2009 samples may reveal more of the impacts this cattle treatment has on SOM properties.

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Effects of Seismic Exploration on Pygmy Rabbits

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ABSTRACT

Pygmy rabbit behavior and above ground burrow characteristics were monitored during seismic exploration in northern Utah in the fall of 2008. Burrow entrance characteristics (height and width) were evaluated at distances up to 250 m from the geophone line before and after the seismic survey. Burrow heights after the seismic survey were significantly lower than pre-treatment measurements ≤ 25 m from the geophone line, but were unchanged at farther distances. Burrow height was reduced by minor sloughing presumably caused by sonic vibrations emitted by vibroseis trucks. Burrow entrances were collapsed if they received a direct hit by a vibroseis tire or shaker pad. Radio collared pygmy rabbits living near the seismic activity were not displaced from their home ranges by the seismic exploration. Vibroseis tracks typically extended an average of 16 m on either side of the geophone line, and most burrow effects were experienced within ~ 10 m of this impact zone. A 50m buffer around known active burrow sites is therefore sufficient to prevent damage to pygmy rabbit burrows by seismic exploration. Further studies are needed to evaluate the effects of seismic exploration on rabbits living in the direct path of seismic activity.

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INTRODUCTION

Oil and gas exploration and development are rapidly expanding worldwide. The process of locating and assessing subterranean oil and gas (termed seismic exploration) consists of mapping of the potential resource field with controlled acoustic energy recorded by a network of receivers (geophones) that are placed along transects, hereafter called geophone lines. Seismic energy transmitters are mounted on large trucks (vibroseis), which generate a vibratory force through a plate that is placed in contact with the ground. Four vibroseis trucks travel abreast on both sides of the geophone line, stopping at regular intervals to transmit vibrations. Dynamite is used to create acoustic energy in locations that are inaccessible to vibroseis trucks. While the influence of oil and gas development on terrestrial wildlife is well studied (for example: Cameron et al. 1992; Lyon and Anderson 2003; Sawyer et al. 2006), the effects of terrestrial seismic exploration activities are little understood.

Seismic exploration has the potential to affect wildlife either by increasing noise and activity around them, or through long-term habitat alteration. The footprint of exploration activities can be quite large (Jorgenson et al. 2010), though the exploration activity itself is

relatively brief (weeks to months). To date, most terrestrial seismic exploration studies have occurred in the tundra, prairies, and forests of far northern latitudes. In the far north, seismic exploration can alter plant community structure, cause soil compaction, and accelerate loss of permafrost (Felix and Reynolds 1989), and these effects can be long-term (Jorgenson et al. 2010). The long-lasting linear remnants of seismic exploration in the arctic have been shown to affect bird distribution and nest success (Ashenurst and Hannon 2008). There is evidence to suggest wildlife may react to seismic activity with elevated metabolic rates (Bradshaw et al. 1998; Reynolds et al. 1986), and the cumulative effects of repeated disturbance of individuals may affect population reproductive rates if exploration is widespread (Bradshaw et al. 1998).

In October 2008 a seismic exploration operation was conducted in the Duck Creek grazing allotment in northern Utah, USA. The route of the survey bisected a site that was part of on-going investigations of pygmy rabbit (*Brachylagus idahoensis*) behavior and ecology. At the time of the exploration, pygmy rabbits were petitioned to be listed under the Endangered Species Act (U. S. Fish and Wildlife Service 2008). In 2010 pygmy rabbits were deemed not warranted for protection under the ESA (U. S. Fish and Wildlife

Service 2010). Pygmy rabbits are associated with dense sagebrush (*Artemisia tridentata* ssp.), and self-created burrow systems (Green and Flinders 1980). While aboveground resources are certainly important for pygmy rabbits, the effects of seismic energy on burrow systems could affect pygmy rabbits by altering burrow architecture, and if severe, trapping them inside collapsed burrows. The objectives of this study were three-fold: 1) to monitor the effects of vibroseis activity burrow entrance architecture; 2) monitor the behavior of radio-collared pygmy rabbits during exploration activities; 3) evaluate the efficacy of a 50-m mitigation buffer.

METHODS

The study was conducted in Rich County, Utah, USA. The site ranged in elevation from 1800 m to 2300 m and consisted of rolling hills with small drainages, some with spring-fed perennial streams. The climate was characteristic of shrub-steppe vegetation types consisting of cold winters, warm summers, and most precipitation falling as winter snow (West and Young 2000). Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) was dominant, with basin big sagebrush (*A. tridentata* ssp. *tridentata*) and low sagebrush (*A. arbuscula*) present at much lower frequencies. Snowberry (*Symphoricarpos oreophilus*) was co-dominant with sagebrush on more mesic aspects. The under-story contained a diverse mix of small shrubs, grasses and forbs, both native and non-native. Land was mixed ownership (Bureau of Land Management and private).

The present study occurred on a 7.3 km (4.5 mi) segment of the seismic route. At the time of the seismic survey, pygmy rabbit investigations had been conducted for several years prior, and were to continue for another several months. As part the ongoing study, 16 adult pygmy rabbits (11 Females and 5 males) were captured at burrow sites in spring 2008, and monitored weekly prior to the seismic exploration project (for details see: Wilson et al. 2011). The geophone line was centered within the area of this existing study (figure 1).

The seismic survey was conducted by CGGVeritas (CGGV, Cedex, France) on 23 and 24 October 2008. Prior to the study, all Federal lands were surveyed for pygmy rabbit burrow activity by a private contractor. As per their agreement with the Bureau of Land Management (BLM), CGGV applied 50-m mitigation

buffers around all known pygmy rabbit burrows found by the contractor. In addition, CGGV agreed to apply a similar 50-m mitigation buffer around the minimum convex hull home ranges of the 16 radio-collared pygmy rabbits located within the study area. No exploration activities were conducted within these buffers. Burrow surveys were not conducted on private land.

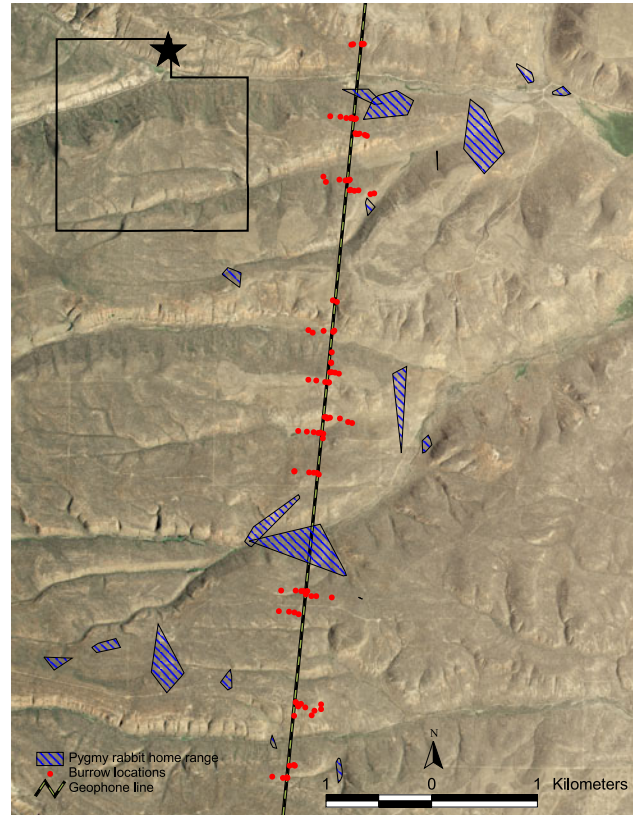


Figure 1. Map of the study area in northern Utah, USA.

Fifteen random vibroseis and five random dynamite locations were selected for burrow measurements. The direction (right or left) of perpendicular burrow transects was randomized based on a coin toss in the field. Burrows were sampled in eight distance classes located along transects at roughly 0, 5, 10, 20, 50, 100, 150, 200, 250 m. In practice burrows weren't always found at every distance class, and all burrows <25 m along the transect were measured. All measured burrows were marked with a metal numbered tag staked near the burrow entrance, flagging tape, and paint so that they could be relocated if collapsed or disturbed during the seismic exploration. Burrows typically enter the ground at an angle; meaning that width and height of the burrow entrance were the most appropriate dimensions for

measurement. Burrow width was measured at the largest point in the horizontal dimension at the mouth of the burrow, and burrow height was measured from the floor to the roof of the burrow opening at the tallest spot. The location of each burrow measurement was marked with blue and orange spray paint to ensure repeated measurements were made from the same location. Burrow measurements were made <1 week prior, and <4 weeks after seismic activity. Paired *t* tests were used to compare the change in burrow dimensions between the pre- and post- seismic measurements. Pygmy rabbits were located visually using homing telemetry immediately prior to and immediately after seismic exploration of the site. Four rabbits were monitored continuously when seismic activity occurred near their home ranges.

RESULTS

None of the rabbits left their home ranges despite the fact that two of them were located within 100m of the geophone line. Other rabbits were located near vibroseis trails (termed snail trails) used by the vibroseis trucks to move between access points on the geophone line. A snail trail on an existing 2-track road bisected the home range of one rabbit. Another rabbit was located near (~120 m) a helicopter landing pad and staging area that was used by CGGV crews for about 2 weeks before and after the survey was conducted on the study area.

Vibroseis vehicles travelled abreast on both sides of the geophone line. The impact zone of the tracks was between 20.7 and 54.8 m (mean = 32.3, standard deviation = 10.5, *n* = 16) wide. Burrow entrances were collapsed if they received a direct hit of a vibroseis truck tire, or shaker plate (*n* = 7). Otherwise, they experienced minor (figure 2), but statistically significant $\mu_D = -2.5$ ($t = -3.080$, $P = 0.004$, $DF = 45$) reductions in burrow height if they were located ≤ 25 m of the geophone line. No change in burrow height was observed for burrows >25m from the geophone line. No changes were observed in burrow width.

DISCUSSION

Pygmy rabbits with minimum convex polygon home ranges ≥ 77 m of the geophone line were not displaced by seismic activity. Before and after measurement of burrows occurring ≤ 250 m of the geophone line indicated that burrows within the

impact zone of the vibroseis trucks (≤ 25 m from the vibrophone line) experienced minor, but statistically significant changes in burrow height. This was presumably due to the vibrations emitted by the vibroseis trucks. A 50-m buffer was an effective mitigation measure for temporary displacement disturbance and from burrow damage by the seismic activity.

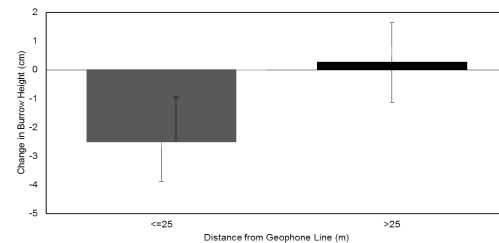


Figure 2. Mean change in height of burrows between the before seismic and after seismic measurements at two distance classes: ≤ 25 m, and > 25 m.

A mean reduction of 2.5 cm in burrow height is not likely to significantly affect the ability of rabbits to use burrows. However, it is not known how deeply the burrows were disturbed. Additional studies are needed to evaluate the impacts of seismic vibrations on the underground portions of burrows. Also, burrow entrances that received a direct hit from a vibroseis truck tire or shaker pad appeared to be collapsed. It is also not known if rabbits potentially trapped inside these collapsed burrows would be able to escape either by using other burrow entrances or digging through the soil and splintered sagebrush blocking the collapsed burrow entrance. The home ranges of all radio-collared pygmy rabbits were excluded from direct disturbance by vibroseis trucks by the 50m mitigation buffer, so it is not known if rabbits living directly in the path of seismic activity would have retreated to a burrow (and thus potentially trapped in a collapsed one), or left the area during activity. These questions should be addressed prior to changing the use of mitigation buffers for pygmy rabbits.

The observed damage to the burrow entrance caused by vibroseis trucks was similar to that caused by sagebrush mechanical treatments. It is common practice when conducting sagebrush treatments to buffer active pygmy rabbit burrows by 50 m. The present study suggests that this buffer distance is

also adequate for seismic exploration. However, it should be noted that there is a difference in the application of mitigation buffers for mechanical treatments and that of the seismic lines. No disturbance is allowed within the mitigation buffer of a treatment, whereas only the geophone line (center line) is mitigated for seismic surveys. Seismic exploration disturbances typically extend 16 m (up to 28 m) on either side of the geophone line. This means that while the width of the actual vibroseis disturbance is ensured to be less than the buffer (and burrows are likely minimally impacted), it does not insure that any vibroseis disturbance is ≥ 50 m from any active burrow. If the intent of applying a mitigation buffer is to insure that there will be at least 50 m between active burrows and the nearest disturbance, then the typical width of vibroseis activity beyond the geophone line should be taken into account when applying mitigation buffers.

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Broom Snakeweed Increase and Dominance in Big Sagebrush Communities

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ABSTRACT

Broom snakeweed (Gutierrezia sarothrae (Pursh) Britt. & Rusby) is a native sub-shrub that is widely distributed on rangelands of western North America. It often increases to near monocultures following disturbance from overgrazing, fire or drought. Propagation is usually pulse driven in wet years, allowing large expanses of even-aged stands to establish and dominate plant communities. It can maintain dominance following fire, or can co-dominate with cheatgrass (Bromus tectorum L.) on degraded sagebrush rangelands. State-and-transition models show that competitive grasses in the respective plant communities can prevent snakeweed dominance.

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INTRODUCTION

Broom snakeweed (*Gutierrezia sarothrae* (Pursh) Britt. & Rusby) is widely distributed across western North America, from Canada south through the plains to west Texas and northern Mexico, and west through the Intermountain region and into California (figure 1). It ranges in elevation between 50 and 2900 m (160 and 9500 ft) and commonly inhabits dry, well-drained, sandy, gravelly or clayey loam soils (Lane 1985). The closely related threadleaf snakeweed (*G. microcephala* (DC) L. Benson) is similar in growth form and appearance, but differs in that it has only 1 to 2 florets per flowering head, compared to 3 to 5 in broom snakeweed. It occurs mostly in the southwest deserts (figure 1).

Broom snakeweed is a native plant that can increase in density when other more desirable plants are reduced or removed by disturbance, such as overgrazing, fire or drought. It can dominate many of the plant communities on western rangelands including: salt-desert-shrub, sagebrush, and pinyon/juniper plant communities of the Intermountain region; short- and mixed-grass prairies of the plains; and mesquite, creosotebush and desert grassland communities of the southwestern deserts (US Forest Service 1937). In addition to its invasive nature, it contains toxins that can cause abortions in livestock (Dollahite and Anthony 1957). Platt (1959) and DiTomaso (2000) ranked it among the most undesirable plants on western rangelands.

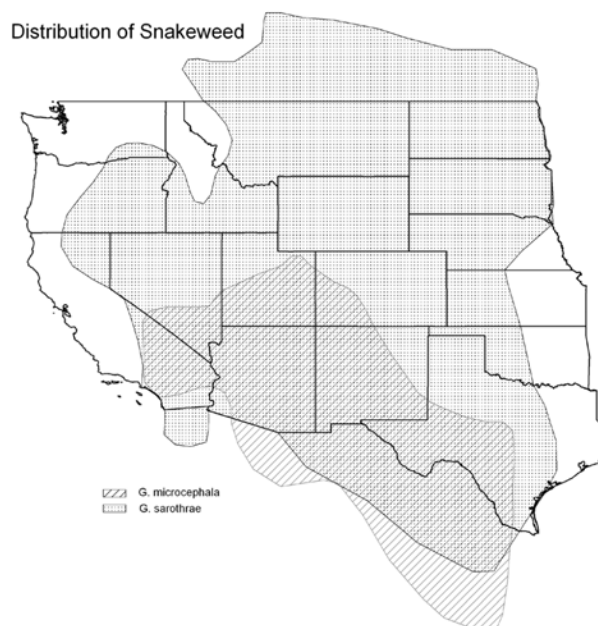


Figure 1. Distribution of broom and threadleaf snakeweed.

ECOLOGY

Broom snakeweed is a suffrutescent sub-shrub, having many unbranched woody stems growing upwards from a basal crown, giving it a broom-shaped appearance. These stems die back each winter and new growth is initiated from the crown in early spring. Once established, snakeweed typically survives 4 to 7 years (Dittberner 1971). It is a prolific seed producer with 2036 to 3928 seeds/plant (Wood et al. 1997). Seeds held in dried flower heads are gradually dispersed over winter. They have no specialized structures such as wings to aid in long

range dispersal, thus they usually drop close to the parent plant. Seeds remain viable into spring, but rapidly disintegrate after May if they remain exposed on the soil surface (Wood et al. 1997).

Germination is light-stimulated (Mayeaux 1983), therefore seeds must remain partially exposed on the soil surface (Mayeux and Leotta 1981). Furthermore, the soil surface must remain near saturation for at least 4 days for the seeds to imbibe and successfully germinate (Wood et al. 1997). Buried seeds remain viable for several years and germinate when moved to the soil surface by disturbance (Mayeux 1989).

Pulse Establishment

The fluctuating resource availability theory of invasibility (Davis et al. 2000) suggests that plant communities are more susceptible to weed invasion whenever there are unused resources. This occurs when there is either an increase in resource supply or a decrease in resource use. Snakeweed populations often establish in years with above average precipitation following disturbance that reduces competition from other vegetation (McDaniel et al. 2000).

Ralphs and Banks (2009) reported a new crop of snakeweed plants ($30/\text{m}^2$) established in a wet spring (precipitation 65 percent above average) in a crested wheatgrass seeding (*Agropyron cristatum* (L.) Gaertner). Intense grazing reduced the grass standing crop (which reduced use of soil moisture by crested wheatgrass) and trampling disturbed the soil surface, thus providing ideal soil and environmental conditions for snakeweed establishment.

In a companion defoliation study (Ralphs 2009), density of snakeweed seedlings was higher in clipped plots in both the crested wheatgrass seeding and in a native bluebunch wheatgrass (*Pseudoroegneria spicata* Pursh) stand. Clipping reduced competition for soil moisture from grass and mature snakeweed plants, allowing new snakeweed seedlings to establish. This study showed that in wet years, snakeweed can establish even in healthy stands of native bluebunch wheatgrass or seeded crested wheatgrass, when defoliation of the grasses reduces competition for soil moisture.

Population Cycles

Pulse establishment allows massive even-aged stands of snakeweed to establish. There is little intraspecific competition among snakeweed seedlings (Thacker et al. 2009a), thus large expanses of even-aged stands establish in wet years. As these stands mature, they become susceptible to die-off, mainly from insect damage or drought stress. Although snakeweed is highly competitive for soil moisture, it is not particularly drought tolerant (Pieper and McDaniel 1989; Wan et al. 1993b). Ralphs and Sanders (2002) reported that snakeweed populations in a salt desert shrub community on the Colorado Plateau died out in 1990, reestablished in 1994, declined in 1996, completely died out in 2000, and have not established during the current region-wide drought (figure 2).

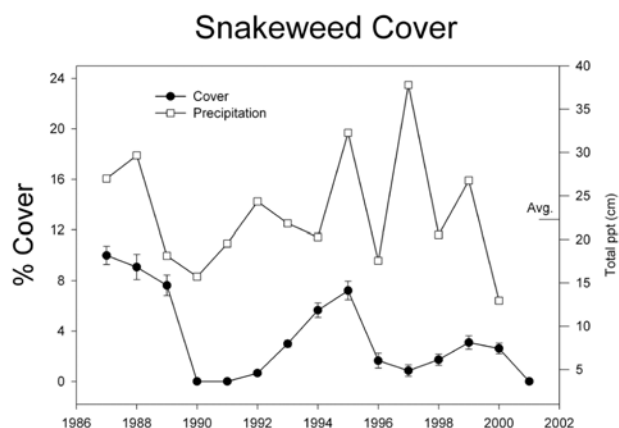


Figure 2. Population cycle of broom snakeweed and annual precipitation.

Competition

Once established, snakeweed is very competitive with other vegetation. McDaniel et al. (1993) reported a negative exponential relationship between snakeweed overstory and grass understory that implies snakeweed's presence, even in minor amounts, suppresses grass growth. Partial removal of snakeweed allowed remaining plants to increase in size and continue to dominate the plant community (Ueckert 1979). Total removal allowed grass production to increase >400 percent on blue grama grasslands (McDaniel et al. 1982, McDaniel and Duncan 1987). Control strategies should strive for total snakeweed control.

Snakeweed's root structure and depth provide a competitive advantage over associated grasses for soil moisture (Torell et al. 2011). In the southwest, its deeper roots enable it to extract soil water at greater depths (30-60 cm), compared to the shallow rooted sand dropseed (*Sporobolus cryptandrus* (Torr.) A. Gray) (Wan et al. 1993c). In its northern range, snakeweed is acclimated to a saturated soil profile from snowmelt and spring rains to sustain rapid growth (Wan et al. 1995). When soil water stress increases seasonally or during drought, leaf stomata do not close completely (Wan et al. 1993a, DePuit and Caldwell 1975), allowing snakeweed to continue transpiring. This depletes soil moisture to the detriment of associated grasses. If drought persists, leaf growth declines and leaves are eventually shed to cope with water stress, but stems continue photosynthesis to enable it to complete flowering and seed production (DePuit and Caldwell 1975). However, as drought stress increases, tissues dehydrate and mortality occurs rapidly (< 10 days) when soil water potential drops below -7.5 MPa and leaf water content declines to 50 percent (Wan et al. 1993b).

State-and-Transition Model

Healthy sagebrush/bunchgrass communities can suppress snakeweed. Thacker et al. (2008) described a fence line contrast between a Wyoming big sagebrush/bluebunch wheatgrass community and a degraded sagebrush/Sandberg bluegrass (*Poa secunda* J. Presl) community in northern Utah. A 2001 wildfire removed the sagebrush in both communities. Snakeweed established on the degraded side of the fence and increased to 30 percent cover and dominated the site by 2005. Bunchgrasses on the other side of the fence prevented establishment of snakeweed.

Thacker et al. (2008) proposed a new broom snakeweed phase to the Upland Gravelly Loam (Wyoming big sagebrush) ecological site state-and-transition model (figure 3) (NRCS 2007). Two "triggers" were identified that lead to snakeweed invasion. Heavy spring grazing over decades eliminated most of the bunchgrass in the plant community, putting the community "at risk" and eventually transitioning from the Current Potential

State (2.2) over a threshold (T2b) to a dense Wyoming Sagebrush State (4). The lack of competition from bunchgrasses allowed snakeweed to establish in the understory. Fire then removed the sagebrush and snakeweed was the first plant to germinate, establish, and rapidly increase and dominate the Snakeweed /Sandberg bluegrass phase (4.2). Subsequent fires will remove snakeweed and the site will likely transition over another threshold (T4b) to a cheatgrass (*Bromus tectorum* L.) community in the Invasive Plant State (5). Thacker et al. (2008) suggests that if robust perennial bunchgrasses can be maintained in the community, they will provide "resilience" to resist snakeweed invasion or expansion, recover from fire or drought, and produce more forage for wildlife and livestock.

CONTROL

Snakeweed can be controlled by herbicides and prescribed burning. McDaniel and Ross (2002) recommended prescribed burning during the early stages of a snakeweed infestation if there is sufficient grass to carry a fire. Herbicide control is recommended on dense snakeweed stands, particularly where fine fuels are not sufficient to carry a fire. Picloram at 0.28 kg ae/ha (0.25 lb/ac) or metsulfuron at 0.03 kg ai/ha (0.43 oz/ac) applied in the fall provided consistent control in New Mexico (McDaniel and Duncan 1987, McDaniel 1989). Sosebee et al. (1982) suggested fall applications were more effective than spring in the southwest because carbohydrate translocation was going down to the crown and roots, thus carrying the herbicide down to the perennating structures. Whitson and Freeburn (1989) recommended picloram at 0.56 kg ae/ha (0.5 lb/ac) and metsulfuron at 0.04 kg ai/ha (0.6 oz/ac) applied in the spring on shortgrass rangelands in Wyoming. In big sagebrush sites in Utah, the new herbicide aminopyralid at 0.12 kg ae/ac (0.11 lb/ac) was effective when applied during the flower stage in fall, as was metsulfuron 0.042 kg ai /ha (1.67 oz/ac) and picloram + 2,4-D at 1.42 kg ae/ha (1.25 lb/ac) (Keyes et al. 2011). Picloram by itself at 0.56 kg ae/ha (0.5 lb/ac) was most effective and eliminated snakeweed when applied in either spring or fall. Residual control was obtained with tebuthiuron (80 percent wettable powder) at 1.1 to 1.7 kg ai/ha (1 to 1.5 lb/ac) on mixed grass prairies in west Texas (Sosebee et al. 1979).

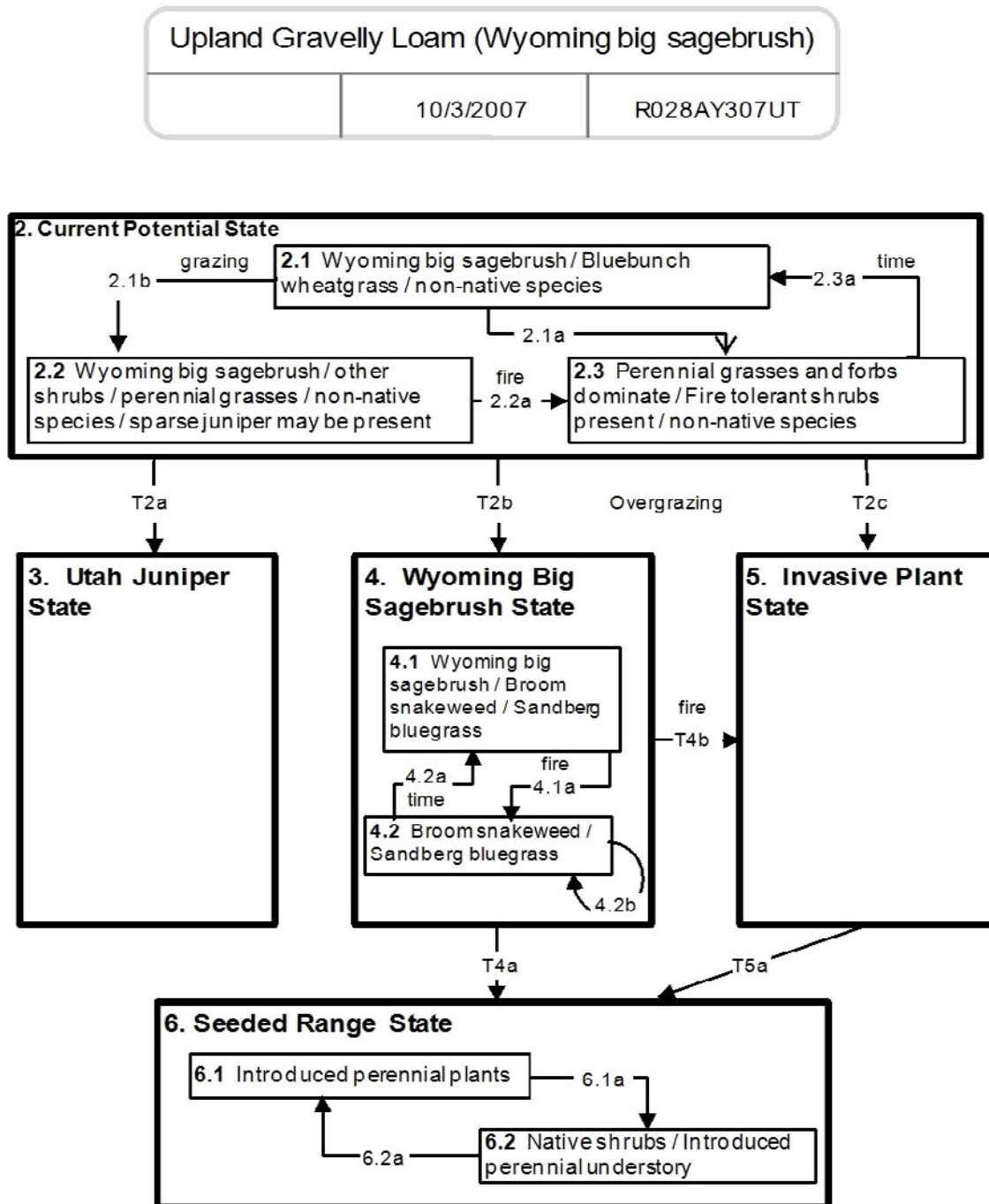


Figure 3. Upland Gravelly Loam (Wyoming big sagebrush) Ecological Site state-and- transition model. Available: <ftp://ftpfc.sc.egov.usda.gov/UT/Range/D28AY307UT.pdf> [Nov 17, 2007].

After snakeweed control, a weed-resistant plant community should be established to prevent reinvasion of snakeweed, cheatgrass and other invasive weeds. Thacker et al. (2009a) reported competition from cool season grasses prevented establishment of snakeweed seedlings in both potted-plant and field studies. Snakeweed seedlings appear to be sensitive to competition from all established vegetation, including cheatgrass. Hycrest crested wheatgrass (*Agropyron cristatum* (L.) Gaertner x *A. desertorum* (Fisch. Ex Link) Schultes) was the most reliable grass to establish on semi-arid rangelands, thus was most effective in suppressing snakeweed establishment and growth (Thacker et al. 2009b). There appears to be a window of opportunity for grasses to suppress snakeweed in its seedling stage, if the grasses can be rapidly established. However, once established, snakeweed is very competitive and will likely remain and dominate the plant community.

SUMMARY

Broom snakeweed is an invasive native sub-shrub that is widely distributed across rangelands of western North America. In addition to its invasive nature, it contains toxins that can cause death and abortions in livestock. It establishes in years of above average precipitation following disturbance by fire, drought or overgrazing. This allows widespread even-aged stands to develop that can dominate plant communities. Although its populations cycle with climatic patterns, it can be a major factor impeding succession of plant communities. Snakeweed can be controlled with prescribed burning and herbicides, however a weed-resistant plant community should be established and/or maintained to prevent its reinvasion.

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Process-Based Management Approaches for Salt Desert Shrublands Dominated by Downy Brome

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ABSTRACT

Downy brome grass (Bromus tectorum L.) invasion has severely altered key ecological processes such as disturbance regimes, soil nutrient cycling, community assembly, and successional pathways in semi-arid Great Basin salt desert shrublands. Restoring the structure and function of these severely altered ecosystems is extremely challenging; however new strategies are emerging that target and attempt to repair ecological processes associated with vegetation change. In this paper, we review the essential processes required to reduce downy brome abundance and assist with creating suitable conditions for revegetation of Great Basin salt desert shrublands.

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INTRODUCTION

Ecosystem processes of Great Basin shrublands have been altered by the persistent effects of past land-use and subsequent invasion of exotic annual plant species (West 1983a, b; Blaisdell and Holmgren 1984; Anderson and Inouye 2001; West et al. 2005). The invasive annual grass downy brome (*Bromus tectorum* L.) is the most notable invasive species in this region. Downy brome dominance is known to alter disturbance regimes, soil nutrient cycling, community assembly, and successional pathways (Belnap et al. 2003; Rimer and Evans 2006; Adair et al. 2008). As an ecosystem driver, downy brome poses serious obstacles to ecosystem resilience and the ability of land managers to repair ecosystem structure and function (Belnap and Phillips 2001; Booth et al. 2003; Chambers et al. 2007).

Restoring ecosystems to pre-disturbance conditions is not always feasible because biotic and abiotic thresholds may have been crossed (King and Hobbs 2006). A pragmatic alternative is to develop management goals to restore key ecosystem properties and processes, including ecosystem resilience (Whisenant 1999; Walker and Langridge 2002). The science of restoration ecology, and the application of ecological restoration to accelerate or initiate ecosystem recovery have emerged in the last few decades (Jordan et al. 1987), and the principles and tools to influence recovery are emerging for damaged Great Basin shrublands (Pickett et al. 1987;

Sheley and Krueger-Mangold 2003; Krueger-Mangold et al. 2006; Sheley et al. 2009b). Collectively, these principles suggest that three critical elements are needed: 1) assess the underlying above and belowground processes responsible for invasive plant dominance (Eviner and Chapin III 2003; Eppstein and Molofsky 2007); 2) develop and apply effective management strategies that affect the causes of invasion and reduce invasive plant dominance (Krueger-Mangold et al. 2006; Sheley et al. 2010); and 3) re-establish native and introduced plant species with appropriate traits to perform well in a restoration setting (Call and Roundy 1991; Jones et al. 2010). This process-based approach requires more than just controlling invasive species, but also actions that influence above and belowground ecological processes (Ehrenfeld 2003, 2004), directly remedy colonization dynamics (Adair et al. 2008), mediate interactions between invasive and desirable species (Eiswerth et al. 2009), and recognize the existence of potential plant-soil feedbacks (Ehrenfeld et al. 2005). A primary challenge facing rangeland management today is to integrate these elements.

ASSESSING ECOLOGICAL PROCESSES

Site assessment seeks to identify a broad array of potentially important ecosystem processes and predict which are likely responsible for continued dominance by invasive plants. These fall into three primary categories: 1) processes that regulate colonization referred to as *site availability*, 2)

processes that regulate the relative abundance of different species termed *species availability*, and 3) the final category consisting of processes regulating the interactions of plants with the above and belowground environment that are referred to as *species performance* (Pickett et al. 1987). Site assessment is a necessary exercise because it reveals how ecological processes are influenced by historical events and the current ecological conditions, and how they can be modified to attain desired ecosystem trajectories and targets (Sheley and Krueger-Mangold 2003; King and Hobbs 2006). Below, we briefly review these three primary categories in reference to salt desert shrublands in the Great Basin.

Site Availability

Historical disturbances are widely recognized as important drivers of invasive plant dominance in Great Basin shrublands. Since colonization by European immigrants in the 1840s, these ecosystems have been used for dryland farming and managed grazing systems, which broadly expanded in response to homesteading acts of 1862-1916 (Gates 1936). The dry farming boom was short-lived and unsustainable in the Great Basin because of the combined effects of low soil moisture and precipitation, changing climate conditions, and soil erosion (Stewart and Hull 1949). Consequently, this practice was largely abandoned; except where climatic conditions and soils matched the requirements of crop species, such as wheat and barley (Young and Evans 1989). Managing these shrublands as grazing systems was also unsustainable, as native grasses and forbs had not evolved with heavy grazing pressure by domesticated ungulates (Mack and Thompson 1982). In addition, native vegetation could not possibly recover from stocking rates and grazing practices that were developed within mesic regions where immigrants had originated. Although grazing intensity has substantially declined in the last 50 years (Piemeisel 1951), the legacy of overgrazing and abandoned farming practices remain today (Jones 2000; Morris and Monaco 2010).

Theoretically, ecosystems that experience novel disturbances are believed to have crossed irreversible thresholds, and will remain in an altered ecosystem state, bounded by current climatic and edaphic conditions (King and Hobbs 2006; Suding and Hobbs 2009). Understanding and characterizing how these

disturbances have altered site conditions and key ecosystem processes has been a major research thrust in the last 20 years (Allen-Diaz and Bartolome 1998; Elmore et al. 2006; Chambers et al. 2007). This research indicates that novel disturbances and altered ecosystem processes within Great Basin shrublands have reduced biological soil crusts, diminished the abundance of native herbaceous species, accelerated soil loss and erosion, and enabled broad scale colonization, spread, and dominance by exotic annual species, foremost among them, downy brome (*Bromus tectorum* L.) (Brandt and Rickard 1994; Young and Longland 1996; Young and Allen 1997; Muscha and Hild 2006).

Exotic annual plant dominance primarily influences site availability by maintaining a disturbance regime that makes it nearly impossible for native species to persist. When abundant, senesced biomass produced by annual species creates a contiguous supply of fine fuel that increases the extent and intensity of fire (Young and Evans 1978; Young and Blank 1995; Brooks et al. 2004). Fire can kill certain shrub species with poorly protected meristems located above ground, including big sagebrush (*Artemisia tridentata* Nutt.) (Ziegenhagen and Miller 2009). In addition, perennial native grasses and forbs can be injured and experience reduced growth and seed production when fire return intervals are decreased (Wright and Klemmedson 1965; West 1994). On the contrary, annual grasses, which complete their life cycle prior to the hot and dry conditions when summer fires occur, are not directly hindered by fire, but their seeds can be diminished by fire, depending on fire dynamics (Sweet et al. 2008; Diamond et al. 2009). Consequently, the fires fueled by annual species favor their further dominance and the subsequent decline in desirable species abundance (D'Antonio and Vitousek 1992; Brooks et al. 2004). Fire frequency in Great Basin shrublands are believed to have increased since European colonization, but this trend has not been fully quantified, and is often implied from historical patterns and indicators (Baker et al. 2009; Mensing et al. 2006). However, in salt desert ecosystems, fire has indeed emerged as a novel disturbance to these low elevation shrublands in the last 30 years (West 1994; Jessop and Anderson 2007; Haubensek et al. 2008).

Mechanistically, disturbance regimes alter site availability through their influence on niches and safe sites for plants and seed (Eckert et al. 1986; Lamont

et al. 1993). For example, disturbance directly modifies competitive interactions (Eliason and Allen 1997), micro environmental conditions (Melgoza et al. 1990; Bradford and Lauenroth 2006), litter dynamics (Sheley et al. 2009b), seed movement (Chambers 2000), and resource supply rates (James and Richards 2007). Characterizing how disturbance influences these processes is an important aspect of clarifying how site availability can be modified by managers to yield a more desired plant community.

Species Availability

Species availability and subsequent colonization depends on propagule dispersal and propagule pressure (Marlette and Anderson 1986; Rodríguez-Gironés et al. 2003; Chytrý et al. 2008). These mechanisms of colonization are critical components of succession because viable seeds must be present through dispersal, from seed banks, or be introduced artificially, as in a rangeland seeding (Call and Roundy 1991; Cox and Anderson 2004). Recent theoretical discussions suggest that colonization dynamics follow certain assembly rules (Ackerly 2003), where both biotic and abiotic filters regulate propagule dispersal and propagule pressure (D'Antonio et al. 2001; Mazzola et al. 2008). In altered shrublands of the Great Basin where disturbances are frequent, colonization is dominated by exotic annual species, which produce abundant seed that dominate seed banks (Humphrey and Schupp 2001). For example, individual plants of downy brome can produce up to 6,000 seeds, most of which will germinate the following fall and rapidly recolonize after disturbance (Smith et al. 2008). In contrast, native perennial grass and shrub species have much slower growth rates and have lower seed output (Young and Evans 1978). Thus, remnant native species experience a highly competitive environment, with reduced fecundity and productivity caused by exotic annual species dominance, which allows it to persist even after earnest control efforts (Borman et al. 1991; Morris et al. 2009).

Assembly rules following disturbance also suggest that priority effects may be responsible for exotic annual species dominance (Tilman 1994; Corbin and D'Antonio 2004; Ludlow 2006). Priority effects describe how exotic annual species achieve greater colonization following disturbance because they often have earlier phenological development, and are more represented in seed banks (Humphrey and Schupp

2001; Rice and Dyer 2001). For example, species that arrive and germinate first can gain dominance and control subsequent community pathways, i.e., successional trajectories (Mack and D'Antonio 1998; Corbin and D'Antonio 2004). Priority effects must be diminished before the performance of desirable perennial species can even be realized. These colonization and species availability obstacles suggest that management actions will need to systematically reduce propagule pressures of invasive species in unison with artificially seeding desirable species and fostering their future dispersal (Corbin and D'Antonio 2004). Furthermore, assessing site conditions will provide critical information about colonization dynamics and indicate potential ways to manipulate species availability when developing a management plan.

Species Performance

There is a robust scientific literature demonstrating functional differences between invasive species and the native species that are negatively impacted by their presence (Vitousek et al. 1997; Ehrenfeld 2003). However, because many factors and processes regulate species performance within an ecosystem, predicting why certain species become invasive, and identifying which ecosystems will be invaded has been challenging (Reichard and Hamilton 1997; Moles et al. 2008). A few of the widely recognized factors important to regulating species performance include resource availability, and the ability of plants to capture resources, ecophysiological traits, plant response to stresses, and tradeoffs in life history traits (James et al. 2010).

The influence of resource availability on plant performance has long been recognized. However, formal theories that seek to explain how resource dynamics regulate relative species competitive ability, species diversity, ecosystem functions, and exotic species invasion are relatively recent (Huenneke et al. 1990; Burke and Grime 1996; Goldberg and Novoplansky 1997; Davis et al. 2000). In general, temporal and spatial aspects of resource capture have emerged as critical components explaining these processes. Annual exotic species perform better under elevated resources for many reasons, including the coincidence of their phenology and temporal resource availability in shrubland ecosystems (Blank 2008). Alternatively, native

perennial species often initiate growth and resource capture after exotic species have pre-empted limiting resources (Melgoza et al. 1990; Chambers et al. 2007). Pre-emption is a consequence of exotic annual species having lower temperature thresholds for root growth (Bradford and Lauenroth 2006), higher nutrient and water uptake rates (Melgoza et al. 1990; Evans et al. 2001), and faster growth rates than native perennial grasses (Arredondo et al. 1998). Thus, without management intervention of ecological processes, invaded sites favor exotic annual species performing at their full biological potential, and their continued dominance.

High exotic annual species performance and dominance on Great Basin shrublands may also be perpetuated by plant-soil feedbacks wherein soil nutrient cycling processes have been altered in ways that primarily benefit annual species (Ehrenfeld and Scott 2001; Evans et al. 2001; Norton et al. 2004; Blank 2008). For example, evidence suggests that downy brome-dominated patches have higher nitrogen mineralization rates, higher total nitrogen availability, abundant low C: N ratio leaf litter, and higher litter decomposition rates than adjacent patches dominated by native species (Evans et al. 2001; Booth et al. 2003; Norton et al. 2004; Rimer and Evans 2006). Not only do these alterations favor downy brome, but they may promote soil organic matter decomposition and further impoverish sites, making them potentially more difficult to rehabilitate with native species (Norton et al. 2004).

Reducing the performance of exotic annual species requires carefully executed management efforts that effectively manipulate the processes responsible for their success while influencing processes that favor desirable species. For example, if site and species availability have been adequately remedied by reducing disturbance frequency and priority effects that favor annual species, the performance of desirable species can be enhanced to trigger different ecosystem assembly patterns where interference from exotic annual species is minimized. Achieving these conditions may be one of the most challenging aspects of land management in salt desert ecosystems dominated by downy brome.

PROCESS-BASED MANAGEMENT

Managing processes has not been the primary objective of land management in the past. For example rangeland managers in grazed semiarid shrubland systems historically adopted the notion that plant communities change linearly toward a climax endpoint dominated by certain late successional species (Clements 1936), and that managers could adjust livestock stocking rates to reverse successional trends (Dyksterhuis 1949). However, this interpretation could not predict non-linear dynamics, or indicate underlying mechanisms responsible for vegetation dynamics (Westoby et al. 1989). Thus, a successional model that incorporates the mechanisms and pathways of succession into a mechanistic framework for process-based management was developed for predicting vegetation change and developing desired changes (Connell and Slatyer 1977; Pickett et al. 1987; Sheley et al. 1996). This model has recently been shown to greatly increase restoration success over traditionally applied integrated weed management (Sheley et al. 2009a), and is gaining credence within rangeland and restoration ecology (Sheley and Denny 2006; Sheley et al. 2007; Sheley and Bates 2008; Sheley et al. 2008). This process-based approach to managing invasive plants advocates assessing site conditions, identifying the ecological processes in need of repair, applying appropriate tools, and re-assessing management outcomes (figure 1; Sheley et al. 2010).

A primary challenge to process-based management is developing the appropriate methods and tools to go beyond treating symptoms of invasive plant problem and begin influencing processes that yield desirable change (Sheley and Krueger-Mangold 2003; Krueger-Mangold et al. 2006). Although, many tools currently exist to remedy invasive annual grass infestations, there is a need for greater understanding of their ability to affect site availability, species availability, and species performance, and whether these tools effectively direct succession to a more desirable vegetative state. Assessing whether potential tools influence the intended ecological processes and yield the desired outcomes is thus necessary to develop predictive, process-based management strategies.

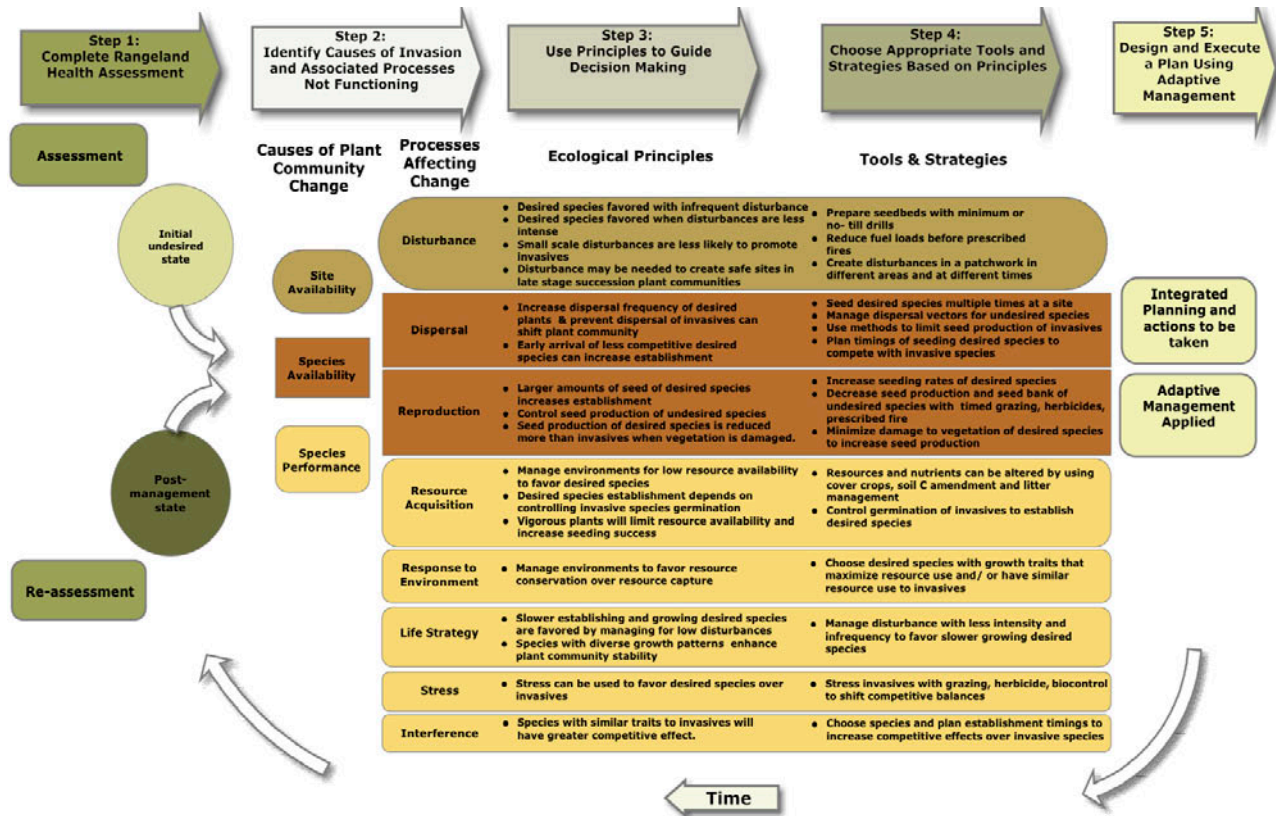


Figure 1. Ecologically based invasive plant management (EBIPM) model (Sheley et al. 2010).

SUMMARY

Process-based management is intended to manage invasive species through targeting the causes of community change. It is likely that no tool alone simultaneously impacts all causes of community change. Therefore, it may be more prudent to use tools in combinations in order to realize the maximum effects. For example, research that evaluates the combined influence of fire, mowing, and pre-emergence herbicides in the Great Basin is currently limited, especially for salt desert shrublands. Quantifying how these integrated tools impact the ecological processes that effect plant community change could help clarify ecological principles, and define improved strategies for annual grass invaded ecosystems in the Great Basin.

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Morphological and Physiological Traits Account for Similar Nitrate Uptake by Crested Wheatgrass and Cheatgrass

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ABSTRACT

Millions of hectares throughout the Intermountain West are either dominated or threatened by the invasive annual grass *Bromus tectorum* (cheatgrass). This invasion is largely linked to disturbance and few regions appear immune. Disturbance liberates resources in a community and cheatgrass appears exceptionally able to capitalize on these resources. One species, however, is consistently competitive with cheatgrass. *Agropyron cristatum* (crested wheatgrass), an improved plant material developed from several populations in central Asia, is drought resistant, grazing tolerant, and largely excludes cheatgrass in stands established within the Great Basin. While previous studies document high resource uptake ability by crested wheatgrass, it remains unknown if high uptake in this species is due to morphological or physiological adaptation. We examined N uptake and tissue morphology of four grasses common in the Intermountain West, including cheatgrass and crested wheatgrass. We also included two native grasses, *Pseudoroegneria spicata* (bluebunch wheatgrass) and *Elymus elymoides* (bottlebrush squirreltail). We observed similar rates of N uptake by cheatgrass and crested wheatgrass and their uptake was greater than the native perennial species. A multivariate analysis suggests that, of the three perennial grasses examined here, crested wheatgrass is morphologically most similar to cheatgrass, but that morphology only accounts for 57 percent of the variation in N uptake capacity among species. Consequently, physiological traits such as induction of N uptake or N efflux likely play a role in the ability of crested wheatgrass to achieve N uptake rates similar to cheatgrass.

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INTRODUCTION

The most substantial plant invasion in North America is the replacement of perennial sagebrush communities with invasive, exotic annual grasses such as cheatgrass (D'Antonio and Vitousek 1992; Chambers et al. 2007). *Bromus tectorum* L., a winter annual native to central Asia, has come to occupy millions of hectares in the Intermountain West over the past century (Stewart and Hull 1949; Bradley and Mustard 2006). This species germinates in the autumn, produces copious seed, and maintains high density stands that rapidly dry near-surface soil in the spring (Knapp 1996; Leffler et al. 2005). Because *B. tectorum* senesces in late spring and seeds can cause livestock injury, the resource value is greatly reduced compared to communities of native perennials (Knapp 1996).

Mounting evidence suggests that plant invasion is largely linked to resource availability and dynamics. Specifically, disturbances cause abrupt increases in resource availability and invasive species are capable

of taking advantage of the resource pulse (Davis et al. 2000; Davis and Pelsor 2001; Leffler and Ryel 2012). Consequently, any disturbance that alters resource availability can trigger an invasion if an appropriate species is present locally. *Bromus tectorum* initially establishes in degraded range communities (Knapp 1996) where perennial grasses are not able to use near-surface soil resources effectively (Leffler and Ryel 2012). When *B. tectorum* reaches sufficient density and fire occurs, remaining perennial vegetation that is not fire-resistant can be eliminated and the site is converted to an annual species plant community (Knapp 1996). In the absence of perennial vegetation, there is abundant soil NO_3^- during autumn in *B. tectorum* stands (Booth et al. 2003; Hooker et al. 2008), which promotes establishment of the next generation because *B. tectorum* is highly responsive to soil N pulses (James 2008).

Since Elton (1958) ecological theory has held that diverse communities of species are resistant to invasion because they more fully occupy niche space, leaving few resources available to be exploited (i.e., niche complementarity, Naeem et al. 2000; Fargione

and Tilman 2005). Conversely, diverse communities are more likely to include a hyper-competitive species that inhibits further invasion (i.e., the sampling effect, Naeem et al. 2000; Fargione and Tilman 2005). In the Intermountain West, few communities seem capable of resisting invasion by *B. tectorum* when they are disturbed; rather one species seems most capable of competing with *B. tectorum*. Crested wheatgrass (*Agropyron cristatum* and closely related species) is an exotic perennial grass planted throughout the region because it is resistant to drought, cold, and grazing (Rogler and Lorenz 1983). This species appears competitive with invasive annual grasses (Rummell 1946; D'Antonio and Vitousek 1992; Davies 2010) and previous studies indicate its ability to acquire soil resources is greater than native grasses (Caldwell et al. 1985; Eissenstat and Caldwell 1988).

The competitive ability of *A. cristatum* is somewhat curious given its perennial life form. Tissue economics theory (Wright et al. 2004) holds that short-lived species will have rapid rates of resource acquisition compared to long-lived species and largely attributes this difference to tissue morphology. *Bromus tectorum* is known to use water and acquire N rapidly when actively growing (Leffler et al. 2005; James 2008), a trait expected in an annual grass. Studies, however, suggest that *A. cristatum* does not have a greater growth rate than native perennial grasses that do not effectively compete with *B. tectorum* (Eissenstat and Caldwell 1987; Bilbrough and Caldwell 1997). Thus, the morphological and physiological mechanisms responsible for interference of *B. tectorum* by *A. cristatum* remain elusive.

In this contribution we address differences in nitrogen uptake capacity and tissue morphology among four grass species grown in two experimental temperature conditions. Grasses include the non-native species *B. tectorum* and *A. cristatum* and the native bunchgrasses *Pseudoroegneria spicata* (bluebunch wheatgrass) and *Elymus elymoides* (bottlebrush squirreltail). We wish to know if (1) nitrogen uptake capacity and tissue morphology differ among species, (2) whether tissue morphology can explain differences in uptake capacity among species, and (3) if differences among species are consistent between growth environments. We conclude with a discussion of the roles morphology and physiology play in competitive ability in these species.

METHODS

Study Species

We compared morphological and physiological differences between the annual grass *Bromus tectorum* L. and three perennial grasses common in the Intermountain West. The perennial grass of most interest was Hycrest II crested wheatgrass (*Agropyron cristatum* [L.] Gaertn.), a widely planted non-native cultivar developed from several populations in central Asia. *Agropyron cristatum* was chosen for its vigorous growth and evidence that rangeland plantings of *A. cristatum* are largely resistant to invasion by annual grasses (Davies 2010). Two native perennial grasses were included in the study for comparison with *A. cristatum*: Rattlesnake bottlebrush squirreltail germplasm (*Elymus elymoides* [Raf.] Swezey ssp. *elymoides*) and Anatone bluebunch wheatgrass germplasm (*Pseudoroegneria spicata* [Pursh] A. Löve ssp. *spicata*). *Elymus elymoides* was selected because it is a short-lived perennial that can naturally establish in annual-dominated ecosystems (Hironaka and Tisdale 1963; Hironaka and Sindelar 1973) while *P. spicata* is a long-lived perennial grass. Seeds of *B. tectorum* were collected from populations in northern Utah. Seeds of the others grasses were obtained from local seed producers.

Growth Conditions, Experimental Treatments, and Measurement

Individual plants were grown in pots (4 x 21 cm 'containers', Ray Leach Inc., Canby, Oregon) for the duration of the experiment. Pots were filled with a 1:1 mixture of a coarse and fine growth medium (Turface MVP and Greens Grade, Profile Products LLC, Buffalo Grove, Illinois) holding ca. 0.5 g H₂O g⁻¹ medium. The medium was washed before use in the experiment to insure no nutrients were adsorbed to the particle surface. Three to five seeds of a single species were added to a pot and kept moist with periodic watering. Ten days following germination and emergence, pots were transferred to growth chambers programmed for constant experimental temperature and a 14/10 h day/night cycle. Photosynthetic flux density inside the chambers was ca. 900 $\mu\text{mol m}^{-2} \text{s}^{-1}$ above the uppermost leaves. After several days of growth, seedlings were thinned to two individuals per pot.

Plants were kept in two growth chambers that each contained twenty pots of each species. One chamber was set to 10 °C, the other to 25 °C. Temperature was monitored (model Watchdog B101, Spectrum Technologies, Plainfield, Illinois), and each week, plants and temperature conditions were rotated among chambers to minimize chamber effects. While target temperatures were achieved at night, daytime temperatures were 3-4 °C warmer than intended (13.1/9.9 and 28.6/24.8 °C day/night, respectively). Moisture in pots was maintained near saturation daily with an NH_4^+ -free nutrient solution containing 0.20 g l⁻¹ KNO₃, 0.21 g l⁻¹ Ca(NO₃)₂, 0.06 g l⁻¹ NaH₂PO₄, 0.12 g l⁻¹ MgSO₄, and 0.3 g l⁻¹ of a complete micronutrient fertilizer (J.R. Peters Inc., Allentown, Pennsylvania).

Nitrogen acquisition was measured via uptake of NO₃⁻, the most available inorganic form of N in semi-arid regions of the Intermountain West (West 1991). Nitrate acquisition was determined with incubations in a ¹⁵N solution (BassariRad et al. 1993). Assays began with the 25 °C individuals ca. 5 weeks following germination. Individuals in the 10°C treatments were measured approximately two weeks later to account for slower development of individuals in the colder temperature treatments. Ten pots of each species were randomly selected for measurement. Individual plants were removed from pots and washed free of growth medium. Eight individual plants were placed in eight flasks each containing 250 mL of 60 atom percent K¹⁵NO₃, the remaining two plants were treated as controls and placed in flasks of K¹⁴NO₃.

After a two-hour incubation at growth temperatures, plants were removed from assay flasks and immersed in a chilled 50 mM KCl solution to stop NO₃⁻ uptake. Plants were then washed five times in distilled water. Root and shoot tissue was dried at 75 °C for at least 48 hours, weighed, and ground to a fine powder using a shaker mill (model 2000, SPEX CertiPrep, Metuchen, New Jersey). Tissue samples were analyzed for [¹⁵N] at the University of California, Davis. Standard deviation among repeated measurements of a standard was less than 0.0003 atom percent. Nitrate acquisition was calculated by determining the difference in tissue ¹⁵N between labeled plants and control plants. Excess ¹⁵N in root and shoot tissue were combined and acquisition is expressed as mass-specific absorption rate (SAR_M, μgN g⁻¹ h⁻¹).

We quantified root and leaf morphological traits before tissue was dried. Surface area of leaf tissue was measured with a leaf area meter (model 3100, Li-Cor Inc., Lincoln, Nebraska) and root images were obtained with a flatbed scanner and analyzed for length and surface area using the software package WinRhizo (Regent Instruments, Quebec, QC). Mass of tissue was determined after drying.

Statistical Analysis

We used ANOVA to test for statistically significant differences among species in each temperature regime. Response variables included SAR_M, leaf area, leaf mass, root area, root mass, and root length in a fixed-effect, one-way ANOVA. Means were separated using a Tukey multiple comparison. Differences were considered significant at $P < 0.05$ and ANOVA was conducted with PROC GLM in SAS (version 9.2, SAS Institute Inc., Cary, North Carolina).

The five morphological variables were combined in a Principle Component Analysis (PCA) to produce new, uncorrelated variables, which could be used in further analyses. We performed PCA using the PRINCOMP function in the statistical computing language R (R Core Development Team 2005). Mass-specific absorption rate was regressed onto the first two principle component axes to determine how much of its variation was explained by tissue morphology.

RESULTS

We observed significant differences among species in SAR_M and all morphological variables in 10 and 25 °C conditions (table 1). The two exotic species (*B. tectorum* and *A. cristatum*) had statistically similar SAR_M as did the two native perennial grasses *P. spicata* and *E. elymoides* (figure 1). Mass-specific absorption rate was 213 percent and 53 percent higher in the exotic species compared to the natives at 10 and 25 °C, respectively.

We found the highest leaf mass in *A. cristatum* and the highest leaf area in *B. tectorum* (figure 2). The two native species were similar in leaf mass and area in both temperature conditions. While differences between *B. tectorum* and *A. cristatum* in leaf mass were significant but small, *B. tectorum* produced ca. double the leaf area of *A. cristatum* at 25 °C. Consequently, *B. tectorum* had a higher specific leaf area (grand means: *B. tectorum* = 210, *A. cristatum* = 91). Higher temperatures resulted in greater leaf mass and leaf surface area for all species.

Table 1. ANOVA examining differences among species in the two temperature conditions. Numerator degrees of freedom = 3 for all variables.

	10 °C			25 °C		
	<i>n</i>	<i>F</i>	<i>P</i>	<i>n</i>	<i>F</i>	<i>P</i>
SAR	32	14.94	< 0.001	31	8.70	<0.001
Leaf Mass	40	34.12	< 0.001	37	29.22	<0.001
Leaf Area	40	99.25	< 0.001	40	256.8	<0.001
Root Length	39	243.3	< 0.001	39	47.01	<0.001
Root Area	40	162.1	<0.001	39	33.20	<0.001
Root Mass	40	32.02	< 0.001	40	8.97	<0.001

Table 2. PCA loadings and proportion of variation explained by each principle component. The first two principle components explain 94% of the variation in the leaf and root morphology data set. The first principle component, a nearly equal weighting of all variables, represents the tissue economics spectrum (Wright 2004).

	PC1	PC2	PC3	PC4	PC5
Leaf Mass	-0.405	0.600	-0.346	0.568	0.182
Leaf Area	-0.472	-0.152	0.774	0.352	-0.178
Root Length	-0.426	-0.577	-0.206	0	0.665
Root Area	-0.483	-0.270	-0.454	-0.115	-0.689
Root Mass	-0.445	0.460	0.184	-0.734	0.134
% explained	0.760	0.179	0.041	0.019	0.002

Root mass was similar in *B. tectorum* and *A. cristatum* but *B. tectorum* produced longer roots and roots with more surface area (figure 3). Consequently, specific root length was highest in *B. tectorum* (grand means: *B. tectorum* = 12.0, *A. cristatum* = 6.8). Root mass, length, and surface area were generally lower in the native perennials. High temperature resulted in greater root mass for all species. High temperature resulted in greater root length and root surface area for the perennial species, but not for *B. tectorum*.

The principle component analysis produced two uncorrelated variables that explained 94 percent of the variation in the morphology data set (table 2). The first principle component (PC1) was a nearly equal weighting of all morphological variables which were correlated with one another. The second principle component (PC2) was more heavily weighted toward leaf mass, root mass, and root length but indicated a

negative relationship between leaf mass and root length. All species and the two temperature conditions separated along PC1 (figure 4). Mean PC1 scores were most similar for the two non-native species and these scores were distinct from those for the native species, which were also similar. *Bromus tectorum* separated from other species along PC2. A regression of SAR_M onto PC1 and PC2 indicated that PC1 explained ca. 57 percent of the variation in SAR_M while PC2 was not a significant predictor (figure 5).

DISCUSSION

Bromus tectorum and *A. cristatum* had similar SAR_M, which was greater than SAR_M of native bunchgrasses. A previous study suggested differences in root length-specific absorption rate among the same species was dependent on N availability; at low N, uptake by native perennials

exceeded that of *A. cristatum* and *B. tectorum* (James 2008). In separate studies, *Agropyron desertorum*, a species closely related to *A. cristatum*, had a greater ability to acquire soil phosphorus (Caldwell et al. 1985) and fix carbon (Caldwell et al. 1981) than native bunchgrasses. Nitrogen uptake by *A. cristatum* can exceed uptake by *B. tectorum* (James 2008) but *B. tectorum* has a higher tissue N concentration (Monaco et al. 2003) and reduces soil N to a greater extent than native grasses (Blank et al. 2010).

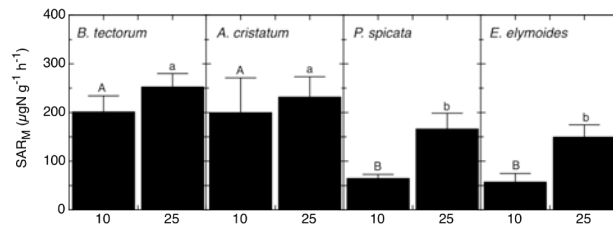


Figure 1. Mean mass-specific absorption rate (SAR_M) under two growth temperatures for each species. Error bars indicate 95% confidence intervals. Bars with the same capital letters above are statistically not different in the 10°C growth condition; bars with the same lowercase letters above are statistically not different in the 25°C growth condition.

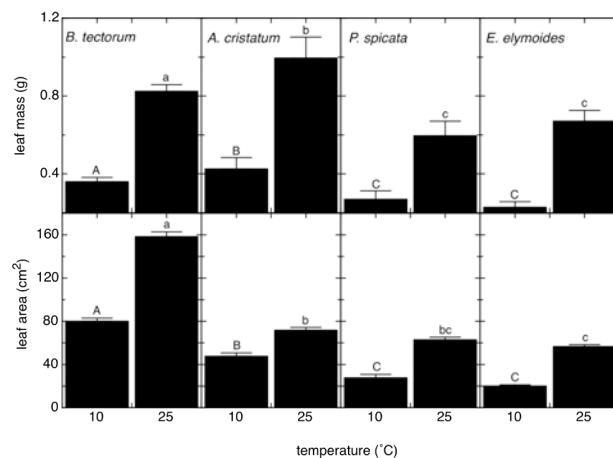


Figure 2. Mean leaf mass and area under two growth temperatures for each species. Error bars indicate 95% confidence intervals. Bars with the same capital letters above are statistically not different in the 10°C growth condition; bars with the same lowercase letters above are statistically not different in the 25°C growth condition.

Tissue morphology differences between an annual grass such as *B. tectorum* and perennial grasses were expected based on tissue economics (Wright et al. 2004). Consequently, *B. tectorum* had high specific leaf area and specific root length, a result previously

reported (James 2008). Most morphological differences, however, were subtle. For variables such as leaf and root mass, *B. tectorum* and *A. cristatum* were not statistically different. For other variables, *A. cristatum* values were intermediate to those for *B. tectorum* and the native grasses. Subtle and context-dependent differences in morphology among these species are common. James (2008) reported greater root mass, root length, and total biomass in *A. cristatum* compared to native perennials, but much greater values for the same variables in *B. tectorum*. Caldwell et al. (1981) attributed higher photosynthetic rates in *A. desertorum* compared to *P. spicata* to higher leaf surface area.

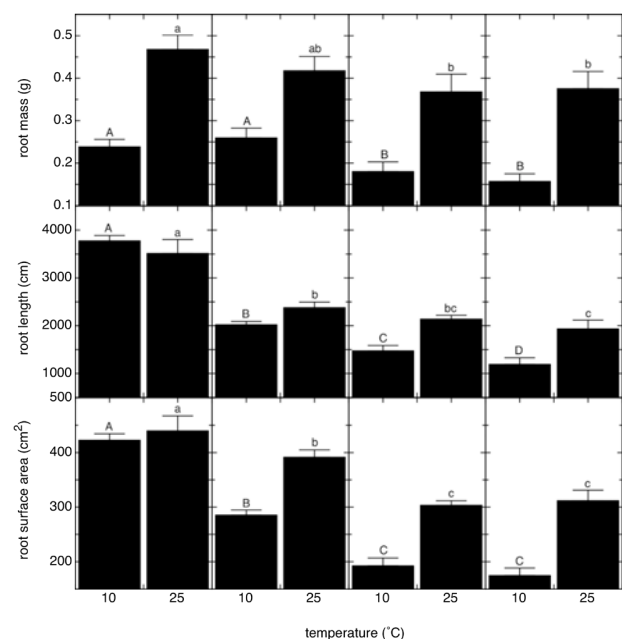


Figure 3. Mean root mass, length, and surface area under two growth temperatures for each species. Error bars indicate 95% confidence intervals. Bars with the same capital letters above are statistically not different in the 10°C growth condition; bars with the same lowercase letters above are statistically not different in the 25°C growth condition.

The first principle component, which accounts for 76 percent of the variation in the data set, describes the tissue economics spectrum for these species. The similar loadings of each morphological variable with the same sign indicate these variables tend to change with each other in a positive manner (i.e., species with high leaf mass also tend to have high root length). Consequently, separation of species along PC1 indicates differences among species based on tissue economics. *Bromus tectorum* and *A. cristatum* were close to each other on the 'acquisitive' end of

the spectrum while *P. spicata* and *E. elymoides* were close to each other on the 'conservative' end of the spectrum. The leaf economics spectrum explained ca. 57 percent of the variation among species in N uptake capacity.

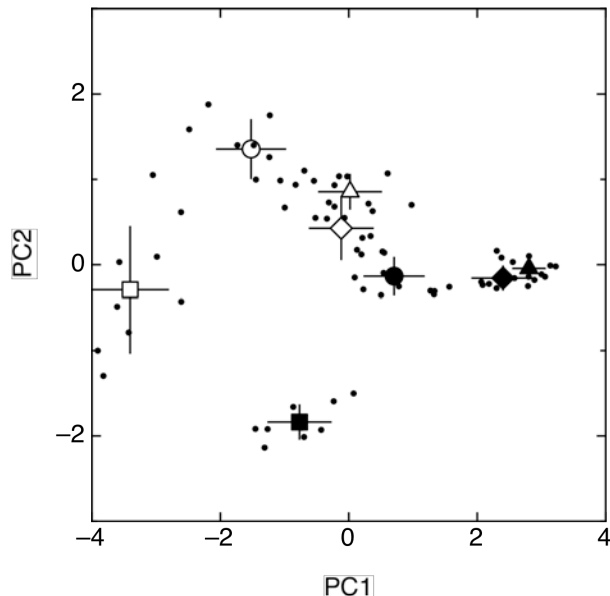


Figure 4. The first two principle components which explain 94% of the variation in the morphology data set. PC1 is a nearly equal weighting of the morphology variables and represents the tissue economics spectrum. Closed symbols represent plants in the 10°C condition while open symbols represent plants in the 25°C condition. Shapes are as follows: square, *B. tectorum*; circle, *A. cristatum*; diamond, *P. spicata*; triangle, *E. elymoides*. Symbols are mean values for a species in a growth condition; error bars represent one standard deviation. Dots indicate scores on PC1 and PC2 for individual plants. Species close to one another on PC1 are similar in morphology.

Growth temperature had little influence on the relationship among species in either N uptake or morphology. Higher temperature clearly shifted species toward the 'acquisitive' end of the tissue economics spectrum, but their position relative to each other on the spectrum did not change. Other growth conditions would likely have a different result, because differences among these species were not consistent when grown in various N environments (James 2008) and *B. tectorum* tends to be very plastic in dry mass production (Rice and Mack 1991).

Our results are specific to the experimental conditions in an N environment that exceed common field values (Booth et al. 2003; Hooker et al. 2008); our goal was

to examine SAR_M when N was not limiting production. Because only 57 percent of the variation in N uptake capacity could be explained by variation in morphology, the remaining variation can be attributed to physiology and measurement error. *Agropyron cristatum* could achieve similar N uptake to *B. tectorum* for several hypothetical physiological reasons: (1) NO_3^- efflux from *A. cristatum* roots could be very low. Our measurements of N uptake quantify a net rate, or a balance between influx and efflux, which is dependent on root $[NO_3^-]$ (Aslam et al. 1996; Glass et al. 2001). Root $[NO_3^-]$ may be related to NO_3^- reduction in leaves or different rates of turnover between influx and efflux systems (Aslam et al. 1996, Glass et al. 2001), which may vary among species; (2) The relative importance of constitutive and inducible NO_3^- uptake systems may differ between *A. cristatum* and *B. tectorum* (Crawford and Glass 1998); and (3) *Bromus tectorum* and *A. cristatum* may have different optimum temperatures for growth which were not measured here. Consequently, at higher or lower temperatures, uptake by *B. tectorum* may exceed that of *A. cristatum* because N uptake and relative growth rate are highly correlated (Glass et al. 2001; Tian et al. 2006). We cannot speculate on how much of the remaining variation is due to physiology; additional studies are necessary to investigate these hypotheses.

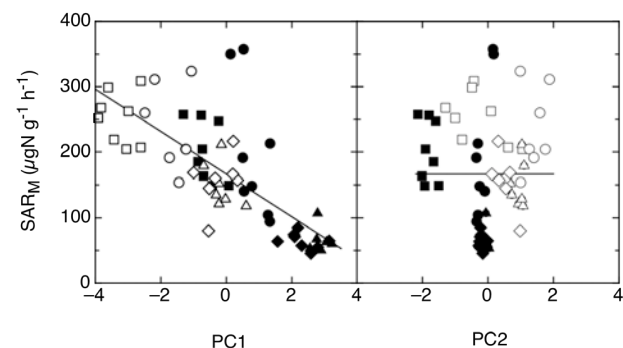


Figure 5. Linear relationship between mass-specific absorption rate (SAR_M) and PC1 or PC2. Symbols as in figure 4. The relationship is significant for PC1 ($n = 63$, $r^2 = 0.57$, $P < 0.001$), but not for PC2.

High N uptake capacity in *A. cristatum* may contribute to the ability of stands of this species to resist invasion by *B. tectorum* and other annual grasses. One difference between annual and perennial communities is the pronounced pulse of N availability in the autumn following senescence of annual grasses (Booth et al. 2003; Hooker et al. 2008). This

pulse of N may be important to establishment of annuals and *A. cristatum* may deny annuals this resource more efficiently than native perennials. Morphological similarity between *B. tectorum* and *A. cristatum* plays an important role, but likely not the only role, in the ability of *A. cristatum* to match N uptake by *B. tectorum*. Future efforts to develop materials for restoration of rangeland degraded by *B. tectorum* should examine temporal N uptake capacity as a desirable trait.

ACKNOWLEDGMENTS

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Evidence that Invasion by Cheatgrass Alters Soil Nitrogen Availability

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ABSTRACT

We hypothesized that cheatgrass (*Bromus tectorum*), an exotic invasive annual, may alter soil nitrogen availability. In the Honey Lake Valley of northeastern, California, we have monitored soil and vegetation along a chronosequence of cheatgrass invasion. In 2007, we measured total C, total N, and $\delta^{15}\text{N}$ in tissue of cheatgrass, winterfat (*Krascheninnikovia lanata*), freckled milkvetch (*Astragalus lentiginosus*), and Indian ricegrass (*Achnatherum hymenoides*) in areas invaded for 1, 4, and >10 years. As time since invasion increased, tissue N increased and C/N decreased significantly for cheatgrass and winterfat. Time since invasion significantly affected $\delta^{15}\text{N}$, which declined significantly for winterfat and increased significantly for Indian ricegrass and freckled milkvetch. These data suggest that cheatgrass invasion has altered soil nitrogen availability and that other plants respond to this altered availability.

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INTRODUCTION

Plants that can engineer the soil or create positive plant-soil feedbacks to enhance nutrient availability can elevate their competitive stature (Ehrenfeld 2003; Kulmatiski and others 2008). Such tipping of competitive stature may be responsible for turning an exotic species into an invasive one (Crooks 2002). The invasive success of *Bromus tectorum* (cheatgrass) is predicated on myriad factors, but soil nutrient availability, particularly of nitrogen (N), is an important determinant (Adair and others 2007; Vasquez and others 2008). The literature is conflicting regarding the effects of cheatgrass invasion on soil N resources. Rimer and Evans (2006) reported that after 2 years invasion by cheatgrass in Canyonlands National Park, Utah, the labile N pool decreased 50 percent. Over a 2-year period, few consistent differences in N mineralization, extractable soil N, or total soil C or N were found between native and cheatgrass invaded sites in Oregon (Svejcar and Sheley 2001). On the other hand, in northern Utah, soil beneath cheatgrass was shown to increase N availability relative to native species (Booth and others 2003).

We have monitored the invasion of a winterfat community in the Honey Lake Valley of northeastern California, beginning in 1998. A systematic measurement of surface soil properties was begun in 2000 utilizing a transect of 13 points, 50 m apart, that extended from the points first invaded by cheatgrass

(1-4) to points not yet invaded (5-13). By 2007, all plots had become invaded by cheatgrass, albeit the most recently invaded only had small sparsely-spaced plants. The chronological resolution of this monitoring program allows detailed information on how cheatgrass has affected soil N dynamics and its relationship to plant N uptake. We hypothesize that cheatgrass invasion alters the availability of soil nitrogen.

MATERIALS AND METHODS

The study was undertaken in the Honey Lake Valley of northeastern California (40° 08' N; 120° 04' W). Since 1998 we have monitored the invasion of a winterfat (*Krascheninnikovia lanata* (Pursh) A. Meeuse & Smit) community by cheatgrass (*Bromus tectorum*). We define invasion to be when small isolated plants of cheatgrass in winterfat canopies expand to fill shrub interspace positions. Soils are uniform throughout the winterfat vegetation zone and are classified as coarse-loamy, Xeric Haplocalcids (Blank 2008). Annual precipitation averages 230 mm. In March of 2000, a transect of 13 sites, 50 m apart, was laid out beginning at the initial focus of cheatgrass invasion (first 4 sites) to areas yet non-invaded. Several times a year, surface soil (0-30 cm) was collected randomly, in interspace microsites, within 5 m of each study plot. In May 8, 2007, following a winter and early spring of below normal

precipitation, we collected total above-ground biomass for the four most common plant species: cheatgrass, winterfat, freckled milkvetch (*Astragalus lentiginosus* Douglas ex Hook.), and Indian ricegrass (*Achnatherum hymenoides* Roem. & Schult. Barkworth). Collected sites were replicated four times in areas separated by at least 50 meters in three invasion zones: invaded by cheatgrass for 1, 4, and >10 years. Plant material was dried for 48 hrs at 60°C, milled, and sent to the Colorado Plateau Stable Isotope Laboratory at University of Northern Arizona for analysis of tissue N and C concentrations and of tissue $\delta^{15}\text{N}$.

All data were normalized as necessary and analyzed by ANOVA with categorical variables invasion class and plant species, using Tukey's Honest Significance Test at the $p \leq 0.05$ level to separate means.

RESULTS AND DISCUSSION

A significant interaction between cheatgrass invasion status and plant species affected plant tissue N concentrations, plant tissue C/N ratios, and values of $\delta^{15}\text{N}$ (figure 1). Tissue N concentrations significantly increased and C/N ratios significantly declined for cheatgrass and winterfat and remained statistically similar for Indian ricegrass and freckled milkvetch with increasing time since cheatgrass invasion. As time since invasion increased, tissue $\delta^{15}\text{N}$ declined significantly for winterfat and increased significantly for Indian ricegrass and freckled milkvetch.

The natural abundance of $\delta^{15}\text{N}$ data lent support to the hypothesis that cheatgrass invasion has altered soil nitrogen availability. If cheatgrass is accessing a more recalcitrant N pool from soil organic matter, which is less available to natives before invasion, then the newly available pool may have a unique $\delta^{15}\text{N}$ signature that will be reflected in plant tissue N of all species (Högberg 1997). Indeed, the three native species tested differed significantly in $\delta^{15}\text{N}$ among invasion classes. Winterfat has an extensive fibrous root system and a deep penetrating taproot. Its $\delta^{15}\text{N}$ tissue signature, when growing without much competition from cheatgrass, was greater than that of cheatgrass which suggests it is partially accessing a different soil N pool, perhaps deeper in the soil profile.

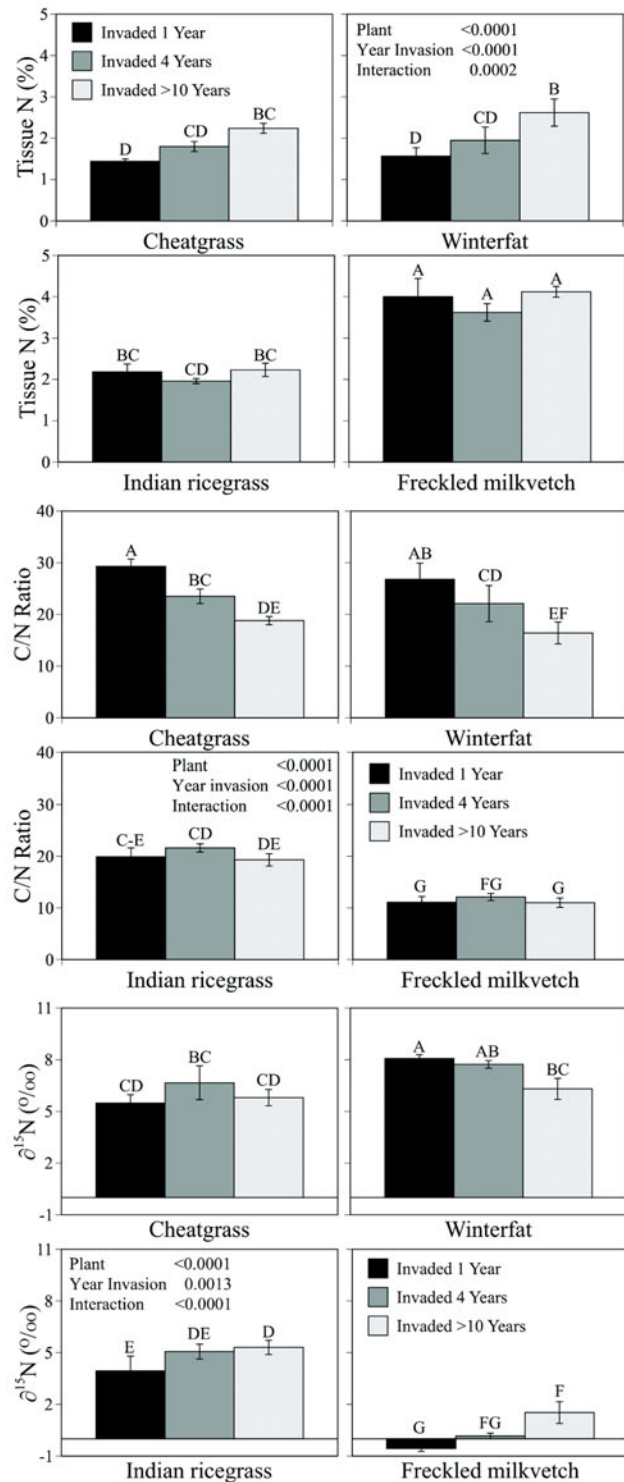


Figure 1. Percent concentration of N, C/N ratios, and $\delta^{15}\text{N}$ in above-ground tissue of cheatgrass, winterfat, Indian ricegrass, and freckled milkvetch as affected by time since cheatgrass invasion. ANOVA results presented in panels. For all panels, bars with different letters are significantly different ($p \leq 0.05$). Data were collected in 2007.

After >10 years of invasion by cheatgrass, $\delta^{15}\text{N}$ of winterfat tissue declined significantly and was statistically similar to that of cheatgrass, which suggests the plant is uptaking a greater proportion of that pool of N associated with cheatgrass invasion. The $\delta^{15}\text{N}$ signature of Indian ricegrass was significantly lower than that of cheatgrass, in plots only recently invaded. Indian ricegrass can fix N_2 in its rhizosheath (Wullstein 1980; Wullstein 1991), which can explain its lower $\delta^{15}\text{N}$. In plots invaded for >10 years, Indian ricegrass tissue $\delta^{15}\text{N}$ significantly increased to an average value similar to that of cheatgrass, again suggesting it may be partially obtaining N resources associated with invasion. The effect of cheatgrass invasion on the $\delta^{15}\text{N}$ signature of the symbiotic nitrogen fixing species, freckled milkvetch, is apparent. Tissue $\delta^{15}\text{N}$ in newly invaded soils averaged near 0 suggesting most N is obtained via fixation of atmospheric N_2 , and increased $\delta^{15}\text{N}$ values in soils invaded for >10 years suggests the plant is using more N from mineralized soil sources after invasion.

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Associations of Near-Surface Soil Moisture and Annual Plant Community Dynamics

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ABSTRACT

*Invasive species have become an increasingly large concern, particularly in already degraded ecosystems, such as sagebrush (*Artemisia tridentata*)-steppe of the Intermountain West. Much of this ecosystem is already infested with large cheatgrass (*Bromus tectorum*) stands and is potentially at risk for future invasions depending on biotic and abiotic conditions. In these ecosystems, the existing vegetation, whether native or non-native, may not effectively utilize the soil moisture resources in the upper portion of the soil, termed the growth pool. If the existing vegetation does not effectively utilize moisture in the growth pool, an open resource is left for the establishment of other plants, including invasives. Through a combination of soil moisture modeling and observational studies, we identified three potential invasion pathways, particularly by annual plants, into a cheatgrass-dominated system, all consistent with the fluctuating resource hypothesis, and all resulting from an available water resource in the growth pool. Results suggest these arid and semi-arid systems are likely to be protected from novel invasive species by complete utilization of growth pool soil water resources by any existing vegetation, whether native or non-native. Our results also suggest the same features which make the site more prone to novel annual invaders may also be useful in guiding establishment of desired vegetation during restoration efforts.*

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INTRODUCTION

The ecohydrology of arid and semi-arid regions can often be characterized by water resources supplied in two forms: a growth pool containing nutrients utilized earlier in the season, typically located in the upper portion (e.g., 0-30 cm depth) of the soil profile, and a maintenance pool to support the transpiration demands of plants growing through the later, and typically hotter, period of the growing season (Ryel and others 2008; Leffler and Ryel 2012). In water-limited ecosystems, effective depletion of soil moisture by established native or non-native vegetation may be the controlling factor in protecting against invasion (Prevéy and others 2010). Failure to utilize the soil moisture resources leaves the system vulnerable to establishment by other plants, either native or exotic.

Several hypotheses to explain plant invasion have been proposed, and some are consistent with the resource pool framework. The empty niche hypothesis considers invasions to be more likely if

species are able to access open resource pools (Mitchell and others 2006). The fluctuating resource hypothesis states invasion is probable when a community either is experiencing a time of unusually high resources, where existing vegetation cannot completely utilize them, or damaged existing vegetation is unable to effectively use normal to high resource levels (Davis and others 2000), provided propagule pressure exists (Davis and others 2000; Chambers and others 2007). The resource-release hypothesis combines the assumptions of resource opportunities in the new environment with release from pathogens or herbivores associated with native habitats (Mitchell and others 2006). The invasion windows may be species-specific or more general (Johnstone 1986). The windows that are species-specific arise through alterations to the biotic or abiotic conditions at a site, as do more general windows (Johnstone 1986). Species-specific windows may also be created through entrance to a community as a seed and establishing once the existing vegetation senesces or is removed through some form of disturbance (Johnstone 1986). In both

frameworks, there is the possibility for invasion failure if invasive propagules are unable to germinate at the time corresponding to open resource pools.

Biotic impoverishment due to land-use practices and invasive plants, particularly annuals, have altered ecosystems in arid and semi-arid areas (Billings, 1990). In the Intermountain West of the US the widespread *Artemisia* sp. (sagebrush)-steppe community has been affected by reduction in the perennial herbaceous community, and dense stands of woody vegetation and novel herbaceous invaders are now present on the landscape (West 1988; Young and Allen 1997). Of particular interest is the invasive annual grass, *Bromus tectorum* L. (cheatgrass) that has altered the soil moisture dynamics of the sagebrush-steppe where it has come to dominate (Kremer and Running 1996; Ryel and others 2010). Cheatgrass is a winter annual, a vegetation type not previously found in the Great Basin (Bradford and Lauenroth 2006). These vegetation changes are linked to altered resource value and ecosystem function.

The stability of these systems prior to anthropogenic alterations may be linked to reduced inter-annual variation in soil water use in the growth pool than is currently found for graminoid and herbaceous rangeland communities in the sagebrush-steppe zone (Ryel and others 2008; Prevey and others 2010). The benefits of system stability as a result of increased species or plant functional type diversity and increased probability of completely utilizing soil moisture resources, arise from biotic interactions among plants (Davis and others 2000; Shea and Chesson 2002) and perhaps the coevolution of species (Thompson 2009) within the original woody-herbaceous communities.

Dense stands of sagebrush effectively deplete both the growth and maintenance pools in most water years (Ryel and others 2010). Given this, sagebrush co-occurring with a cheatgrass understory should provide some degree of protection against novel invaders, even in years when cheatgrass fails to become established. However, cheatgrass monocultures subject to periodic establishment failures may be at high invasion risk, since they essentially revert to bare ground in these years.

We conducted an observational and modeling study to assess whether more complete utilization of the

upper soil moisture contained in the growth pool in a degraded sagebrush-steppe acted to reduce the potential for novel annual invaders. A goal of the study was to identify possible pathways for establishment of an exotic species, all consistent with the availability of sufficient resource pools. An invasion was considered likely if there was an unusually large open resource pool present in the system, with or without damage to the existing vegetation. Since both warm and cool season plants occurred near the study site, we were able to assess the importance of the timing of the open resource pool.

METHODS

Site Description

The field site was located in Rush Valley in west-central Utah (112°28'W, 40°17'N, and elevation 1,600 m). Vegetation types include large patches of near monocultures of cheatgrass (*Bromus tectorum*), big sagebrush (*Artemisia tridentata*) and crested wheatgrass (*Agropyron desertorum*); some big sagebrush stands are growing in association with or bordering cheatgrass, crested wheatgrass or other native perennial tussock grasses. The study area is grazed by cattle each spring.

Soils at the site are silt-loam to over 3.0 m depth. The climate is temperate with cold winters and hot summers. Mean annual precipitation at nearby Vernon, Utah is 240 mm and mean annual temperature is 8.3° C (Ryel and others 2002). The period of temperatures sufficient for plant growth ranges from late March through late October. Soil moisture is recharged mainly by accumulating snowmelt in early spring (Ryel and others 2010); the few summer rains are typically not sufficient to recharge moisture via infiltration to depths greater than 0.1 m (Ryel and others 2003, 2004).

Field Measurements

Measurements were conducted within three patches of vegetation. These included two patches of cheatgrass (~0.5-1.8 ha) and a field of crested wheatgrass (~80 ha). Cheatgrass established at the site following fire in 1992 (Hooker and Stark 2008). The cheatgrass plots are bordered in places by largely monotypic stands of big sagebrush (~5-60 ha). Crested wheatgrass was planted in 1992 and has

remained largely a monoculture since. The portion of the field used in this study has not been grazed since 1999. In spring 2002, an herbicide (Roundup, Scotts Company LLC, Marysville, Ohio) was applied to kill all vegetation within two 10x10m plots. These plots were compared with the undisturbed crested wheatgrass within 30 m of the plot edges.

Soil moisture was measured using two methods. Individually calibrated screen-cage thermocouple psychrometers (J.R.D. Merrill Specialty Equipment, Logan, Utah and Wescor, Logan, Utah) were installed at nine depths through the profile from 30 cm to 300 cm in March, 1999 in cheatgrass, sagebrush and crested wheatgrass monocultures; measurements were collected nearly continuously from spring 1999 to 2003 and converted to volumetric soil water content as in Ryel and others (2002). For 2007-2009, soil moisture data in cheatgrass monocultures were collected with a capacitance probe (model Diviner 2000, Sentek Technologies, Stepney SA, Australia) with soil cores taken periodically for comparison; cores were used to determine volumetric water content from measured gravimetric water content as described in Ivans and others (2003).

Community composition data were collected using 50-m line transects through relatively homogeneous stands or vegetation patches. Species were identified every 0.5 m along four transects and species cover was estimated as the portion of points that contained each species.

In spring 2008, an area dominated annually by cheatgrass with scattered sagebrush experienced a failure in the cheatgrass crop. That spring, sagebrush seedlings established in the bare spaces between mature sagebrush, which had in prior years been dominated by cheatgrass. Sagebrush seedlings were counted and tagged in a 10 m x 15 m plot after this event and monitored for survival in fall 2009 and summer 2010.

Simulation Modeling

Soil moisture, root growth and uptake, and soil hydraulic conductivity were simulated for spring through fall 2000-2002 and 2007-2009 using HYDRUS 1-D (Šimůnek and others 2008). Inputs to the model include soil hydraulic properties (table 1) determined for our study area (Ryel and others, 2002), root distribution, and root water uptake rates. Temperature, precipitation, and potential

evapotranspiration data were obtained from the Cooperative Observer Program (COOP) monitoring station in Tooele, Utah. Based on depth of water extraction (Ryel and others 2010; Ryel, unpublished data), roots of cheatgrass and halogeton were assumed to be limited to the top 45 cm of the soil, while Russian thistle and pepperweed roots were assumed to tap moisture stored deeper in the profile, at 90-120 cm. Root growth and uptake for each vegetation type were initiated at the start of the appropriate growing season for each species. Root growth was specified as daily values to allow for rapid cheatgrass root growth to 45 cm once the upper portions of the growth pool are depleted. The root water uptake parameters (table 2) were estimated from psychrometer measurements for cheatgrass and pepperweed (Ryel and others 2010; Ryel unpublished) and Diviner 2000 and soil core data for Russian thistle and halogeton.

Table 1. Soil hydraulic parameters used in Hydrus 1-D to simulate soil water dynamics and root water uptake. The hydraulic parameters are from the soil catalog (Carsel and Parrish 1988) loaded in Hydrus 1-D for silt-loam.

Soil depth	300 cm
θ_r	0.067
θ_s	0.45
α	0.02
n	1.41 cm^{-1}
K_s	10.8 cm/day

The model was run for March 15-November 10 and was initiated with the soil at 25 percent volumetric water content (field capacity). In 2001 and 2008 the site experienced extensive establishments of novel exotic annuals and minimal cheatgrass cover. In 2009 an eruption of halogeton occurred, following a normal cheatgrass life cycle. All other years in the period 2000-2010 experienced normal cheatgrass establishment and are treated as near monocultures.

The model was also run for 2008 for what would have been a mixed sagebrush and cheatgrass stand had cheatgrass not experienced germination failure. Sagebrush roots were limited to the top 160 cm of the soil (Ryel and others, 2002). Root water uptake between March 15 and November 10 was modeled for only the shrub component. Sagebrush establishment was represented in the model by adding to the root distribution in the upper soil layers.

Table 2. Root water uptake parameters for use in Hydrus-1D estimated for the vegetation types at our site in pressure head (cm). Root water uptake occurs between P0 and P3, with a maximum at P0pt, where it is assumed that root water uptake ceases at soil pressure heads below P3. P2H and P2L are the pressure heads where the roots begin to be limited in their water uptake assuming a transpiration rate of r2H and r2L (cm/day).

Model Parameter	Cheatgrass	Pepperweed	Russian thistle	Halogeton
P0	-15 cm	-15 cm	-15 cm	-15 cm
P0pt	-546 cm	-546 cm	-546 cm	-546 cm
P2H	-920 cm	-1500 cm	-1500 cm	-1500 cm
P2L	-3783 cm	-1800 cm	-4500 cm	-4500 cm
P3	-9102 cm	-9102 cm	-9102 cm	-9102 cm
r2H	0.7 cm/day	0.5 cm/day	0.7 cm/day	0.7 cm/day
r2L	0.1 cm/day	0.1 cm/day	0.1 cm/day	0.1 cm/day

RESULTS

Crested Wheatgrass

The two crested wheatgrass plots subject to herbicide application in early spring 2002 were invaded by halogeton (*Halogeton glomeratus*) during the 2002 growing season. In September 2002, 53 halogeton individuals had established in one plot and 17 in the other (figure 1). No halogeton plants were found outside the two removal plots in the surrounding undisturbed crested wheatgrass stand (> 10 ha), indicating the invasions were limited by existing, undisturbed crested wheatgrass plants. As a summer annual, the halogeton in the plots became established after the crested wheatgrass had been removed and were still green and succulent in September (figure 1).

Sagebrush Establishment

Following the cheatgrass crop failure in fall 2001, 214 sagebrush seedlings established in spring 2008 within a 10 m x 15 m plot where none had established in the previous 10 years. When the plot was re-surveyed in fall 2009 only one of the seedlings had died. Vegetation in the plot in summer 2010 was mature big sagebrush, the new, establishing big sagebrush, and cheatgrass in the spaces between the mature and newly established sagebrush plants.

Vegetation Composition

Although the species composition, especially of the minor members, has changed over the course of our study period, the cheatgrass dynamics are the most

important (table 3). The early germination and failure of cheatgrass predisposed the system to novel annual, spring-germinating invasive plants in 2001 and 2008 (table 3). Cheatgrass re-established as a near-monoculture in spring 2002 and 2009.



Figure 1. The extent of the invasion in two crested wheatgrass plots treated with herbicide in early spring 2002. The pictures were taken on September 26, 2002. The plots had 17 (upper) and 53 (lower) halogeton plants.

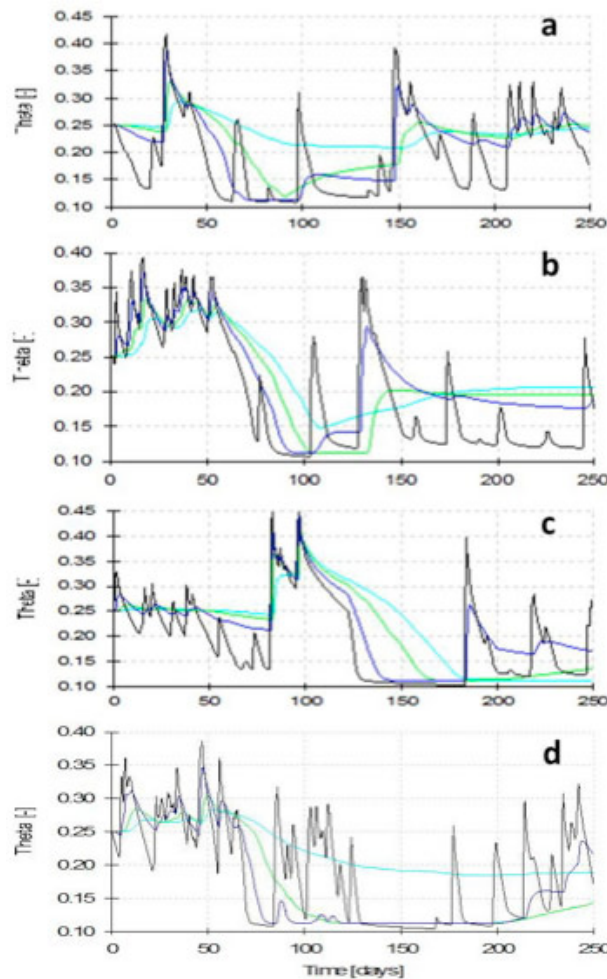


Figure 2. Modeled volumetric soil moisture (θ) trends over the simulated period March 1 (day 1)-November 6 (day 250). Soil moisture in the 3 cm (black), 15 cm (blue), 30 cm (green), and 45 cm (light blue) layers are shown. a. 2007, a normal cheatgrass year. b. 2001, a year of pepperweed eruption in the spring. c. 2008, a year with a July Russian thistle eruption; the large spike on the graph at day 100 corresponds to the start of Russian thistle establishment. d. 2009, a year with halogeton establishment in June following cheatgrass senescence in late May.

Community Invasion Pathways

Soil moisture dynamics in the growth pool were modeled in the cheatgrass community for four years of interest. The model was run for the period March to early November (figure 2). In a typical growing season when cheatgrass was dominant (figure 2a), volumetric water content was 11 percent at the time of senescence at day 90. Although the growth pool was

recharged to 30 cm depth in early August, no further vegetation established at the site during the summer.

A different dynamic was seen in 2001 and 2008 following the fall failure of cheatgrass establishment. In 2001, an eruption of a cool-season novel invader, pepperweed (*Lepidium perfoliatum*) occurred. The pattern of recharge of the growth pool from snowmelt and spring rains and subsequent soil moisture depletion in the growth pool by the pepperweed plants was similar to the water dynamics seen for a typical year dominated by cheatgrass (figure 2a, 2b). No further community changes were found at the site in that year despite a sizable recharge event in mid-July (figure 2b). Some recharge of soil moisture occurred from this event after the cool-season plants senesced as the only water losses from the soil was through evaporation, in the uppermost portion of the soil column.

The growth pool water dynamics in 2008 over the growing season of March-November were much different when the site experienced an eruption of the invasive exotic forb Russian thistle (*Salsola kali*) (figure 2c). As in 2001, the site was nearly bare ground in spring with very minimal cheatgrass cover in early spring, but this continued until mid-summer. The uppermost portion of the soil experienced evaporative losses, but the rest of the profile below 5 cm remained close to field capacity (25 percent water content) until Russian thistle was observed at the site in July following two summer storms. The already large growth pool was added to after the recharge events (figure 2c).

An anomalous June soil moisture recharge event occurred in 2009 (figure 2d), precipitating a substantial halogeton invasion during the rest of the summer. However, by the following spring 2010, the site reverted to a cheatgrass-dominated community. Unlike the situation in 2008, which had water remaining from overwinter recharge (figure 2c), limited water was available below 15 cm and the infiltration from the large event was only sufficient to recharge the uppermost layers of the growth pool (figure 2d).

As an example of the invasibility of bare ground at the site, a simulation was run for bare ground subject to 2007 environmental conditions (figure 3). This year was chosen because it was one of the driest years during our study period. While the top layer of the

growth pool was subject to evaporative losses and recharge, the moisture content in all layers of the growth pool remained higher in the absence any plants drawing down the water content (figure 3). This higher moisture content over the growing season demonstrates the increase in invasion potential.

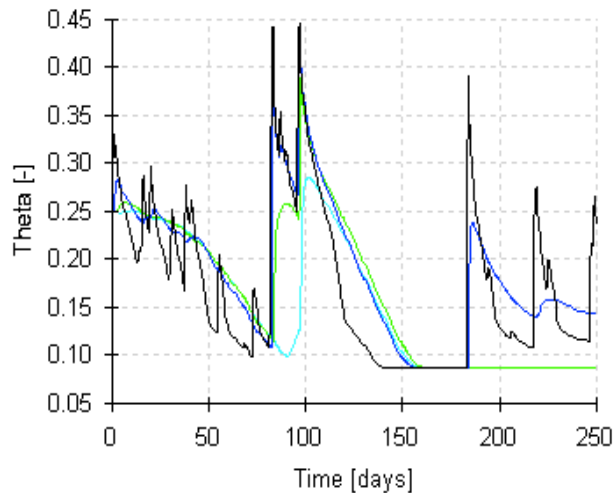


Figure 3. Modeled volumetric water content over the simulation period March 1 (day 1)-November 6 (day 250) for 2007 for bare ground. Soil moisture in the 3 cm (black), 15 cm (blue), 30 cm (green), and 45 cm (light blue) layers are shown.

DISCUSSION

Our results show the importance of the utilization of the growth pool to reduce invisibility, and are consistent with findings in Chambers and others (2007) where cheatgrass invisibility in sagebrush-steppe systems was found to occur when soil moisture was available. In 2008, an unusually large open resource led to invasion by Russian thistle in mid to late July (figure 2c). Community composition surveys in 2001 suggest pepperweed erupted in the system in late spring, during a period of naturally high resource availability. June 2009 was an unusually wet month and this created a resource for establishment of a novel invader during the warm season, a time when this system would be expected to have a reduced risk of invasion. Vegetation dynamics in the cheatgrass stands have experienced rapid community assembly and disassembly during our study period (table 3).

We have identified four pathways into these systems for a novel invader, all consistent with the fluctuating resource hypothesis of Davis and others (2000). Of the other possible hypotheses for invasions, we find limited support for only the empty niche hypothesis

and the invasion windows hypothesis (Mitchell and others, 2006; Johnstone, 1986), but neither of these frameworks can explain all invasion types we observed. The first invasion in the cheatgrass community during our study period in 2001 (figure 2b) was an example of the natural vulnerability of these arid and semi-arid sites where soil moisture recharge to depth is driven by large precipitation events constrained to a single season. Spring, following snowmelt, is a time of higher risk of invasion, given adequate propagule pressure. In the spring water in the growth pool, the most limiting resource, is available, even if the species involved have overlapping root water uptake strategies (Funk and others, 2008). The pepperweed eruption occurred in the spring, after snowmelt, a time of naturally high resources in this ecosystem (figure 2b). The second type of invasion tended to follow soil moisture availability enhanced by cheatgrass crop failure and a sizable early summer precipitation event (figure 2c). This type of invasion combines the two parts of the fluctuating resource hypothesis: disturbance to the existing vegetation and resource addition to the system. The difference between 2001 and 2008 may have been the dryer upper layers in spring and early summer in 2008 that did not favor germination of cool season species such as pepperweed.

The third type of invasion (figure 2d) emerged in 2009, where a normal cheatgrass lifecycle occurred, full germination with senescence in May; halogeton erupted following the recharge event to the shallow portions of the growth pool. In both 2008 and 2009, germination occurred subsequent to precipitation events that resulted in soil moisture that exceeded field capacity near the surface. The crested wheatgrass roundup plots also demonstrate this pathway (figure 1) with unused resources exploited by a summer annual. With disturbance to the existing vegetation, the soil resource created by the winter recharge only experienced evaporative losses, mostly from the uppermost portions of the growth pool. This is the part of the growth pool most likely to be recharged by sizable summer rains at our site. In all of these cases, an open soil moisture resource in the growth pool corresponded to the establishment of novel invaders. Regardless of the community composition, in all years except 2008, the existing vegetation has the ability to draw down the water in the upper layers of the growth pool to nearly the same level, around 11 percent volumetric water content (figure 2a-d).

Table 3. Annual plant community assembly and disassembly has been rapid as seen through yearly changes in species presence. When known, species names are given; otherwise community members are referred to by functional group or family. Plants are listed if one individual has been seen at the site. The community dominant is noted as (d) indicating >90 % of the vegetation cover. Other species had <10% of the vegetation cover.

2000	2001	2002	2007	2008	2009
Cheatgrass (<i>Bromus tectorum</i>) (d)	Pepperweed (<i>Lepidium perfoliatum</i>) (d)	Cheatgrass (<i>Bromus tectorum</i>) (d)	Cheatgrass (<i>Bromus tectorum</i>) (d)	Russian thistle (<i>Salsola kali</i>) (d)	Cheatgrass (<i>Bromus tectorum</i>) (d)
Forbs	Cheatgrass (<i>Bromus tectorum</i>)	Pepperweed (<i>Lepidium perfoliatum</i>)	Bur buttercup (<i>Ceratocephala testiculata</i>)	Cheatgrass (<i>Bromus tectorum</i>)	Scarlet globemallow (<i>Sphaeralcea coccinea</i>)
Cryptobiotic crust	Grass	Grass			Annual wheatgrass (<i>Eremopyrum triticeum</i>)
	Dandelion (<i>Taraxacum</i> spp.)	Dandelion (<i>Taraxacum</i> spp.)			Dyer's woad (<i>Isatis tinctoria</i>)
	Pink geranium (<i>Geranium</i> spp.)	Pink geranium (<i>Geranium</i> spp.)			Halogeton (<i>Halogeton glomeratus</i>) (d, after cheatgrass senescence)
	Bur buttercup (<i>Ceratocephala testiculata</i>)	Bur buttercup (<i>Ceratocephala testiculata</i>)			Bur buttercup (<i>Ceratocephala testiculata</i>)

'Ecological filtering' by existing vegetation can be done on many different environmental factors. George and Bazzaz (1999) found the fern understory to alter the light climate and soil litter depth across various sites in mesic hardwood forests, thus affecting seedling recruitment of some light-sensitive tree species. They found the tree species most likely to establish were those tolerant to reduced light conditions early in their seedling phases. Cheatgrass may act as an 'ecological filter', both in the cheatgrass monocultures and in areas where it occurs as the understory in association with sagebrush. In the monocultures, our results show the effect of removal of cheatgrass, coupled with open resources, on rapid community assembly and disassembly from year to year (table 3; figure 2). In areas where it grows in association with sagebrush, the filtering effect acts to prevent any of the yearly sagebrush seed rain from establishing until disturbance to the cheatgrass eliminates the filtering effect. Cheatgrass, like other plants, modifies its environment, in part through altering the soil moisture dynamics, drawing down the growth pool early in the growing season to levels where nutrient diffusion becomes limited (Ryel and others 2010). This may severely limit germination, growth and establishment of other species that germinate in spring or summer.

Convincingly explaining the mechanisms of past invasions is a significant but important challenge (Davis and Pelsor 2001). With the help of longer-term data sets, knowledge of the limiting conditions at our site, and a vegetationally simple site, we have reconstructed possible mechanisms linked to observed events. These mechanisms linked to invasion dynamics and rapid community assembly were related to ecohydrological dynamics, although, the evolutionary history of the invading species was likely also important. Given that the invaders were biome-shifting Eurasian species (Crisp and others 2009), additional Eurasian invaders, particularly annuals, with the evolutionary history required to effectively utilize water resources in arid or semi-arid regions would be expected to be possible future invaders.

Our work suggests the same features which make the site more prone to novel annual invaders may also be useful in guiding establishment of desired vegetation during restoration efforts. This includes the potential for developing new management strategies for dealing with these, and potentially other, invasive species based around the managing of resource pools (Leffler and Ryel 2012). In particular, arid and semi-arid systems are likely to be protected in large

part from novel invasives by complete or near full utilization of growth pool soil water resources by existing vegetation, whether native or non-native. Management should be directed toward minimizing the opportunities for invasion by minimizing the availability of the growth pools to undesirable species.

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Predicting the Impact of Climate Change on Cheat Grass (*Bromus tectorum*) Invasibility for Northern Utah: A GIS and Remote Sensing Approach

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ABSTRACT

Cheat grass (Bromus tectorum) invasibility represents a serious threat to natural ecosystems dominated by sagebrush (Artemisia tridentata). Ecosystem susceptibility to annual grass invasion seems to be driven by specific biophysical conditions. The study was conducted in Rich County, Utah, where cheat grass invasion is not yet an apparent problem, but an imminent invasion might be just a matter of time (temporal scale) to meet spatial variations in environmental conditions (spatial scale). Literature review and expert knowledge were used to define biophysical variables and their respective suitability ranges of where cheat grass takeover might occur. GIS, remote sensing and logistic regression-statistical analyses were employed to estimate probability of cheat grass invasion along environmental gradients. GIS procedures were used to spatially predict areas prone to be invaded by cheat grass under present climatic conditions (model prediction power was 47 percent). Afterwards, simulated climatic change projections (for 2099 year) from the Community Climatic System Model (CCSM-3) were used to model the invasibility risk of cheat grass. The 2099 cheat grass prediction map showed a favorable reduction of around 25 percent in the areas affected by cheat grass invasion, assuming that climate changes occurred as predicted by the CCSM model. The location of highly predisposed areas can be useful to alert managers and define where resources might be allocated to reduce a potential invasion and preserve native rangeland ecosystems.

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RESUMEN

El riesgo de invasión de Bromus tectorum representa una grave amenaza para los ecosistemas naturales dominadas por Artemisia (Artemisia tridentata). La susceptibilidad del ecosistema a la invasión anual de este pasto parece ser impulsada por condiciones biofísicas espaciales. El estudio se realizó en el Condado Rich, estado de Utah, donde la invasión de esta especie no es aún un problema aparente, pero una invasión inminente podría ser sólo una cuestión de tiempo (escala temporal) para satisfacer las variaciones espaciales en las condiciones ambientales (escala espacial). Revisión de literatura y el conocimiento de expertos se utilizaron para definir las variables biofísicas la adaptabilidad del pasto. Análisis SIG y teledetección y un análisis de regresión logística se emplearon para estimar la probabilidad de invasión a lo largo de gradientes ambientales. Procedimientos SIG fueron utilizados para predecir espacialmente las zonas propensas a ser invadidas por dicho pasto, bajo las condiciones climáticas actuales (2009) (la precisión del modelo fue de 47 percent). Posteriormente, proyecciones simuladas del cambio climático (para el año 2099) del Modelo del Sistema de la Comunidad Climática (CCSM-3) se utilizaron para modelar el riesgo invasibilidad del pasto. El mapa del 2099 mostró una reducción de alrededor del 25 percent de las áreas afectadas por Bromus tectorum, asumiendo que los cambios climáticos ocurren como predice el modelo CCSM. La ubicación de las zonas predisuestas a la invasión pueden ser útiles para alertar a los administradores y definir los recursos para reducir una posible invasión y preservar los ecosistemas nativos.

INTRODUCTION

Cheat grass (*Bromus tectorum*) arrived from Europe more than a hundred years ago and now it has

spread out all over the western US in more than 11 states (Lloyd 1955, West 1999). It can be found in more than 60 millions acres of public and private lands (Wisdom et al. 2005). In the Great Basin desert,

it is estimated that cheat grass already covers around 3.3 million acres (Wisdom et al. 2005). The land management implications of invading cheat grass include the loss of prime wildlife habitat, impact to the regrowth of native vegetation following wildland fire events, soil erosion, loss of rangeland health, and the distribution and expansion of other noxious weeds (Harris 1967, Holechek et al. 1989, Lloyd 1955).

Cheat grass invasibility seems to be driven by genetic conditions, intrinsic to the species, and specific biophysical conditions (Mack and Pyke 1983). Cheat grass has a prolific capacity to produce seeds (Suring et al. 2005). It is able to germinate in the fall or spring, is highly tolerant to recurrent fires and to current grazing practices (Chambers et al. 2007, Pellant 1990). Cheat grass also prepares the site conditions to favor its growth and spread rate. After initial fires, for instance, it increases further risk of subsequent, more frequent fires. This brings serious consequences in terms of loss of wildlife and fish habitat, soil erosion and sedimentation and biodiversity (Bradley and Mustard, 2006). Regarding the biophysical conditions, cheat grass tolerates a wide range of climatic and edaphic conditions (Novak and Mack 2001). Land managers are currently seeking to understand its genetic patterns and preferred biophysical conditions (Bradley et al. 2003).

Invasive species may increase as the climate changes through time (Kriticos et al. 2003). Most the world has already experienced substantial increases in temperature and precipitation as a part of the global climate change scenario (Community Climate System Model project 2010, Morris et al. 2002). Subsequent changes in species distribution, either exotic or native, are expected (Higgins et al. 2003). Managers from federal and state agencies recognize the need of using preventive management to forecast species adaptability and new distributions (Bradley and Mustard 2006).

According to Reichler (2009), Utah will experience a substantial increase in temperature and a decrease in precipitation as a part of the global climate change scenario. Northern Utah is expected to have an approximately 10 percent increase in winter precipitation and a 10 percent decrease in summer precipitation. In general it is expected that this area will receive a uniform warming of ~3°F in winter and ~4°F in summer. According to the same source, other climatic changes will include: less snow pack in

winter, earlier snow melt in spring and in summer, warming will increase water demand and therefore there will be less water flowing from watersheds. Changes in current climate regimes will allow some species to expand their range, while others may be restricted to a narrow range, showing so far many sources of uncertainty (Higgins et al. 2003). To our knowledge, no other efforts have been made to assess ecological changes in cheat grass distribution given a hypothetical scenario of global climate change in Northern Utah using a GIS/remote sensing approach.

The proposed research questions for this study were:

Does cheat grass represent a threat in Rich County, Utah?

If it does, where are the areas prone to be invaded spatially located?

What are the environmental variables that favor cheat grass establishment?

Will there be any change in its spread as a result of an expected climate change?

METHODS

Study Area

The study area was located in Rich County, Utah (figure 1). The area presents an elevation gradient from 1,500 to 2,100 meters above sea level, from East to West. Precipitation places the area in a semi-arid zone, receiving from 200 to 300 mm per year and temperature will usually range between -40 degrees C to 40 degrees C.

The rangelands of Rich County in Northern Utah are largely characterized by having vegetation dominated by sagebrush (*Artemisia* spp.) with associated native and introduced grasses (Shultz 2009), salt desert scrub and pinyon-juniper ecosystems, and other major vegetation types (Washington-Allen et al. 2004). Rich County is best characterized as a higher elevation big sagebrush-steppe / shrubland environment ranging from the pinyon-juniper ecosystems to sub-alpine forests and meadows. These areas have been under commercial agriculture, and grazing for years.

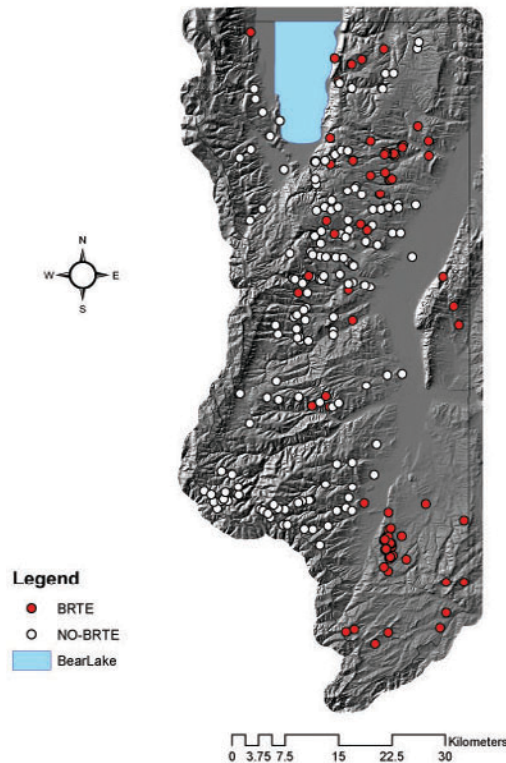


Figure 1. Sampled sites for Cheatgrass (dark color) and Non-cheatgrass sites (white) in Rich Co., UT.

Some big sagebrush ecosystems have converted to exotic annual grasslands or to pinyon-juniper dominance, while an equal area has maintained its natural condition (West 1999). Within shrub-steppe, dominant shrub species included Wyoming big sagebrush (*A. t. wyomingensis*), mountain big sagebrush (*A. t. vaseyana*), basin big sage (*A. t. tridentata*), black sage (*A. nova*), antelope bitterbrush (*Purshia tridentata*), snowberry (*Symphoricarpos* spp.), Utah serviceberry (*Amelanchier utahensis*), rubber rabbitbrush (*Ericameria nauseosus*) and yellow rabbitbrush (*Chrysothamnus viscidiflorus*) (Stringham 2010). Perennial forbs and annual grasses are established following mechanical land treatments to alter woody species abundance and continued heavy livestock grazing. With continued impacts from heavy livestock grazing and mechanical removal of native shrubs, the native grass component is markedly decreased. This plant community is characterized by some grazing tolerant herbaceous species, including cheat grass.

Methodology

Current Scenario 2009

Field data were acquired in summer of 2007. Field forms were developed in a Microsoft Access database to record GPS coordinates and photos of

field sampling locations. A total of 286 field samples were collected from different sources: 50 percent cheat grass, 50 percent no-cheat grass samples. The 143 samples of non-cheat grass sites were taken mostly from the Southwest GAP Analysis project (Lowry et al. 2005). The cheat grass samples were collected by the main author of this paper (S. Rivera), by the T. Edwards' Lab at USU (Edwards and Howe 2009) and by USU RS/GIS Laboratories (Peterson et al. 2008). These data were used as field-input data in these analyzes. Data layers were produced by clipping raw data layers to a 1 km buffered Rich County boundary, and then scaling by standard deviation. The standard deviations were multiplied by 100 and rounded to the nearest whole number. Spatial data was manipulated using ArcGIS ver 9.2, and environmental data was extracted (drilling) from each layer and the R software was used to study potential relationships, linearity, normality and redundancy among variables.

Table 1 shows all explanatory variables used in this study. Most remote sensing derived data were obtained from a Landsat TM scenes taken in 2006. Data manipulation and analyzes were done mostly using the software Erdas Imagine version 8.5. All layers and data points were arranged in ArcGIS ver 9.2 GIS software. Data overlapping and sampling ("drilling"); the xy points into the layers were used in Arc GIS using the sampling function in the spatial analysis toolbox. The Raster calculator was used to draw the spatial distribution based on the resulting logistic model.

Scenario 2099 (A2)

The climate change A2 scenario is considered the worst case scenario if the current world's policies continue and no special actions are taking to combat global warming or environmental change issues (Morris et al. 2002). Climate change projections have been developed by the Community Climate System Model (CCSM-3) on a Gaussian grid, which is commonly used in scientific modeling (Community Climate System Model project 2010). We selected these GIS layers for northern Utah for total annual precipitation (ppt) and average temperature (ta) for 2099 (Thornton and Wilhelmi 2010). Currently, the datasets can be downloaded in a GIS *shapefile* format, where each point represents a centroid of a corresponding CCSM grid cell (IPCC 2007).

Table 1. List of potential explanatory variables used in this study.

Variable	Explanation
Aspect	Aspect, as computed by ArcMap [-1 = flat]
Elevation	Elevation from the USGS National Elevation Data Set (m).
Normalized Difference Vegetation Index (NDVI)	Reflectance at peak, sampling points selected from non-anthropogenic influence sites, Mean annual NDVI changes over the years for a particular site, a composite of maximum.
Slope curvature	Curvature from r_ned_dem calculated by ArcMap (positive values=convex slope, negative values=concave slope)
Northness	Northing coordinate, NAD83, Zone 12Y UTM coordinates (meters)
Eastness	Easting coordinate, NAD83, Zone 12X UTM coordinates (meters)
Slope	Slope from elevation data set (degrees)
Solar flux index	Annual average solar flux calculated using Zimmerman solar radiation model on r_ned_dem and using Daymet monthly temperature grids (kJ/sq.m/day).
Slope contributing area	log of upslope contributing area calculated using Tarboton "Tau DEM" ArcMap plug-in (ln(m))
Relative humidity	Average annual relative humidity grids calculated from Daymet grids (ranging from 0-100%).
Land form	The 10 landform classes were from 1 to 10: 1) Valley flats, 2) Gently sloping toe slopes, 3) Gently sloping ridges, fans and hills, 4) Nearly level terraces and plateaus, 5) Very moist steep slopes, 6) Moderately moist steep slopes, 7) Moderately dry steep slopes, 8) Very dry steep slopes, 9) Cool aspect scarps, cliffs and canyons, and 10) Hot aspect scarps, cliffs and canyons (Manis et al. 2001).
Temperature	Average annual temperature calculated from Daymet grids (1/100 C).
Precipitation	Sum of annual precipitation grids calculated from Daymet grids (1/100 cm)

Both temperature and precipitation files were downloaded from the CCSM data site (Hoar and Nychka 2008) and then data were clipped using the Rich county shapefile and re-projected. We ran a Kriging interpolation analysis to calculate the temperature layers, the average annual temperatures based on the monthly average temperature. For the precipitation file, a new field was created to calculate the sum of the monthly precipitations to obtain the total annual precipitation. The Kriging method utilized was the Universal method with a linear with linear drift semivariogram model (Gebhardt 2003).

It is important to mention that climate models like these are not like weather forecast models. They do not project specific events at the exact time these events occur (like the 1997 El Niño). The CCSM control runs are designed to show internal model variability, by having fixed external forcing. They are more random and statistical representation of such events rather than actual (Community Climate System Model project 2010).

Sampling

All cheat grass and non-cheat grass events or point data sampling was conducted in all 13 layers variables described in Table 1. The *Sample* spatial

analysis function of Arc GIS ver. 9.2 was used to conduct the “drilling” of all layers. The re-sampling algorithm used when re-sampling these raster layers was the nearest neighbor assignment.

Logistic Regression Model

Logistic regression has been used to predict the absence or presence of a particular species (Austin 1985, Dixit and Geevan 2002). A logistic regression model was developed, extracting the information from the “drilling” process in ArcGIS ver. 9.2 using the raster calculator function. The logistic regression model is as follow (equation 1):

$$P = \frac{e^{a + bX}}{1 + e^{a + bX}}$$

OR

$$P = \frac{1}{1 + e^{(\beta_0 + \beta_1 X_1 + \beta_2 X_2 + \dots + \beta_k X_k)}}$$

Equation 1. Logistic regression model.

Where β_0 is a constant and β_i are coefficients of the predictor variables. The computed value, P , is a probability between 0 to 1. This logistic model LM (generalized linear model GLM) was used to simulate

the present/absence of studied species (Fielding and Bell 1997). The presence of cheat grass was considered a success or 1, and the absence a failure or 0.

Model Accuracy

In thematic mapping from geo-referenced data, the term accuracy is used typically to express the degree of 'correctness' of the predicting model (Foody 2002, Gilbert et al. 2005). Model accuracy assessment was performed in this study to compute the probability of error for the cheat grass prediction map (2009). Samples were "drilled" into the final prediction map to determine which samples fell correctly into the modeled classes (Lowry et al. 2008). In the 2009 prediction map: 50 percent was taken as the cut off number. Below 50 percent was considered as an absence and values higher than 50 percent were considered as presence values. A total of 69 samples (20 percent of all samples) were previously withheld randomly for the accuracy assessment. Procedure involved the use of Arc GIS ver 9.2 and the spatial analysis tool: sampling.

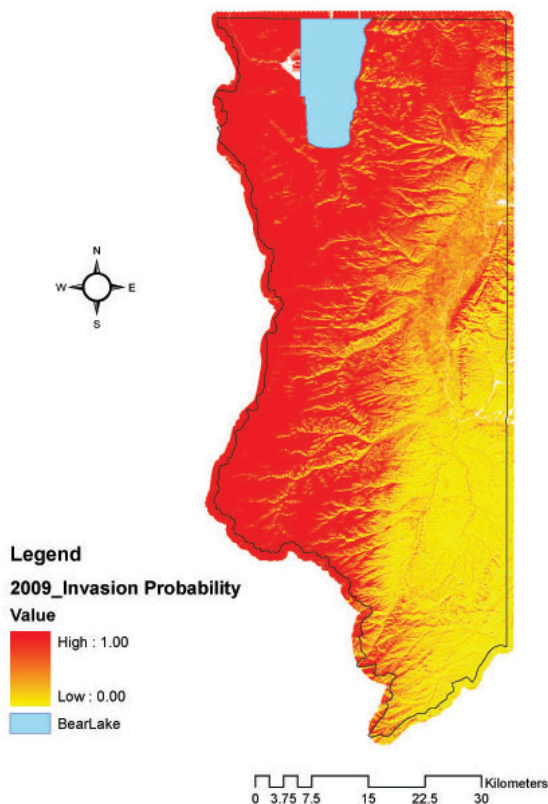


Figure 2. Distribution of 2009-cheat grass and non cheat grass sampling points along the Precipitation 2009 (1/100cm) gradient, Rich County, Utah.

RESULTS AND DISCUSSION

Decrease of Cheat Grass Invaded Areas

Final results showed that there is a decrease of around 20 percent in the 2009 cheat grass invasibility map (figure 2) when compared to the 2009 cheat grass invasibility map. In this case, we observed that the speed of propagation of this invasive species is being restricted by the climatic conditions that are predicted for the 2009-A2 scenario and other studies (Sardinero 2000). In other words, less precipitation, higher temperatures can produce a stress in plant species and reduce the presence of certain species.

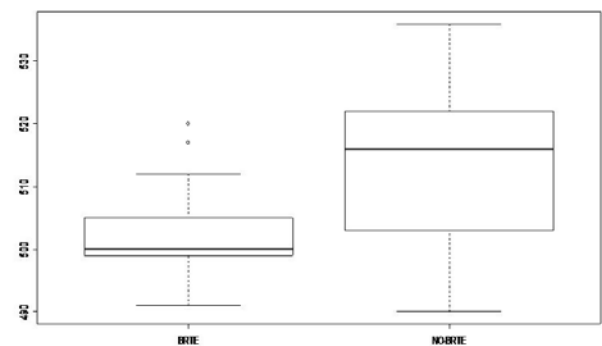


Figure 3. Distribution of 2009-cheat grass and non cheat grass along the NDVI in Rich County, Utah.

Significant Environmental Variables

Final results showed that current (2009) cheat grass distribution in the rangelands in Rich County, may be driven by elevation ($\alpha = 0.001$), solar flux index ($\alpha = 0.001$), relative humidity ($\alpha = 0.001$) and temperature ($\alpha = 0.001$). Slope contributing area also showed some statistical significance ($\alpha = 0.1$) (table 2).

Results of logistic regression analyzes of climate change scenario for cheat grass prediction model in 2009 are shown in Table 3. The highly significant variables were: elevation ($\alpha = 0.001$), solar flux index ($\alpha = 0.001$), temperature 2009 ($\alpha = 0.001$) and precipitation 2009 ($\alpha = 0.001$). The land form category also showed some statistical significance ($\alpha = 0.1$).

In the logistic regression (figure 3), the final model was statistically significant for the following environmental variables: precipitation, temperature, slope contributing area, NDVI and solar radiation. All studied variables and their relationships with the shrub species are described below:

Precipitation: The main driver of presence was humidity at each site. Figure 4 shows that the cheat grass sites receive smaller amounts of precipitation: These sites are generally located at lower elevations.

NDVI: The Normalized Difference Vegetation Index is an indicator of the amount of greenness reflected by the vegetation. Figure 5 shows that the cheat grass sites had lower greenness values when compared with the other plant species.

Relative humidity: Is a measure of atmospheric moisture availability at each site. Figure 3 shows that cheat grass sampling sites showed a lower relative humidity compared with the other types of vegetation.

Elevation: Cheat grass samples were found at lower altitudes between 2,000 and 2,100 meters above sea level, whereas other species were generally found at higher elevations (figure 6).

Table 2. Results of logistic regression analyzes of climate change scenario for the 2009-cheat grass prediction model.

Variable	Statistical significance
Aspect	
Elevation	*** ($\alpha = 0.001$)
Slope curvature	
Northness	
Eastness	
Slope	
Solar flux index	*** ($\alpha = 0.001$)
Slope contributing area	. ($\alpha = 0.1$)
Land form	
Relative humidity	*** ($\alpha = 0.001$)
Temperature 2009	*** ($\alpha = 0.001$)
Precipitation 2009	

These results are very consistent with the literature findings that cheat grass invasibility varies across elevation gradients and appears to be closely related to temperature at higher elevations and soil water availability at lower elevations (Chambers et al. 2007). In addition, the environmental variables identified as significant were consistent with qualitative requirements of the cheat grass's habitat

characteristics. This agreement makes this study comparable to other studies of predicting the invasion of exotic weeds (Collingham 2000). By knowing this, a high agreement between environmental variables, values and species requirements may increase the power of forecasting potential invasions as described by Gilbert (2005).

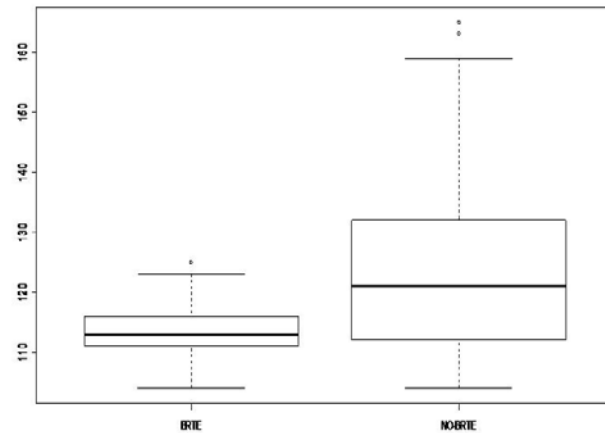


Figure 4. Distribution of 2009-cheat grass and non cheat grass along the Relative Humidity in Rich County, Utah.

Model Validation

The overall accuracy for the 2009 cheat grass distribution model was 31 percent; 46.9 percent for the cheat grass (BRTE) sites and 16.7 percent for the non cheat grass (NO-BRTE) sites (table 4: the confusion matrix and the overall classification accuracy). This indicates that from all withheld sites 47 percent of the cheat grass sites fell correctly into that class in the predicted model. The second analyzed class; non cheat grass species had only 17 percent accuracy. In general, the model performed better at predicting the cheat grass sites. The model also identified a clear and logical distribution pattern along the environmental gradients of elevation, temperature and precipitation. A visual validation was also performed using expert knowledge and field observations. Final distribution was corroborated by experts (Shultz 2010, personal communication) that agreed that final distribution satisfies observed natural distribution tendencies.

The 2009 prediction model could not be validated, since there is no current tool to conduct a validation into a future land cover model.

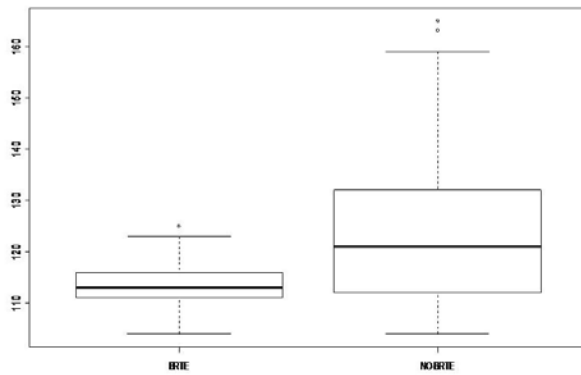


Figure 5. Distribution of 2009-cheat grass and non cheat grass along the elevation (meters) gradient in Rich County, Utah.

Table 3. Results of logistic regression analyzes of climate change scenario for the 2009-cheat grass prediction model.

Variable	Statistical significance
Aspect	
Elevation	*** ($\alpha = 0.001$)
Normalized Difference Vegetation Index (NDVI)	n/a
Slope curvature	
Northness	
Eastness	
Slope	
Solar flux index	*** ($\alpha = 0.001$)
Slope contributing area	
Land form	. ($\alpha = 0.1$)
Relative humidity	n/a
Temperature 2009	*** ($\alpha = 0.001$)
Precipitation 2009	*** ($\alpha = 0.001$)

CONCLUSIONS

Our data indicate that the main driving factors on cheat grass invasion under present conditions are: elevation, temperature, precipitation, NDVI, and relative humidity (figure 7). We can also conclude that under the expected changes in climatic conditions cheat grass establishment will be favored, agreeing literature on analyzing cheat grass propagation and expansion in the Intermountain West, over the past several decades (Bradley et al. 2003, Chambers et al. 2007).

Our data also indicate that the main driving factors on cheat grass invasion under the climate change conditions of scenario A-2, 2099 are: elevation, temperature, precipitation, and relative humidity. In general, again wetter and warmer climatic conditions favor cheat grass establishment, confirming the finding of previous studies (Young and Clements 2007) and personal observations (Shultz 2009) which characterized cheat grass as an opportunistic species.

Table 4. Error matrix of the 2009- cheat grass prediction model and reference data.

Predicted Data	Reference data	
	BRTE	NO-BRTE
BRTE	46.9%	83.3%
NO-BRTE	63.1%	16.7%
% per specie	46.9%	16.7

Overall classification = 31%

It is important to mention that this modeling only predicts cheat grass invasibility based on future climatic condition and does not take into account the probable increase of fires or any changes in management strategies, especially grazing, whose combined effect could potentially trigger a cheat grass spread. The combined effect of fire and grazing, which implies the reduction in of native species, has been identified as significant factors for the growth and reproduction of cheat grass (Chambers et al. 2007).

This study demonstrates the effective use of GIS and remote sensing tools to describe and predict potentially spatial changes in vegetation at the landscape level. Older modeling prediction techniques provided little spatial information of where plant species distribution could be expected to be located in heterogeneous landscapes. GIS and Remote Sensing techniques combined with statistical analyzes, offer a promising tool to place plant distributions along environmental gradients, and thus providing important knowledge of where management efforts might be efficiently directed to mitigate the negative aspects of such possible vegetation change.

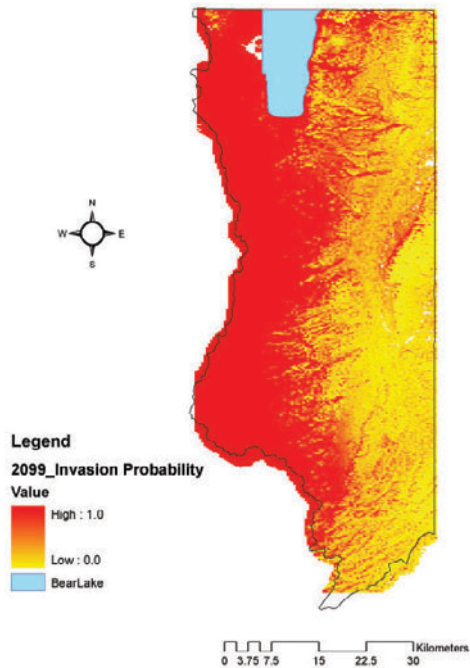


Figure 6. Map of the 2099-cheat grass invasibility model in Rich County, Utah.

ACKNOWLEDGEMENTS

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Potential Impacts of Energy Development on Shrublands in Western North America

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ABSTRACT

Impending rapid development of the abundant energy resources found in western North America may have dramatic consequences for its terrestrial ecosystems. We used lease and license data to provide an approximate estimate of direct and indirect potential impacts from renewable and non-renewable energy development on each of five major terrestrial ecosystems and completed more detailed analyses for shrubland ecosystems. We found that energy development could impact up to 21 percent (96 million ha) of the five major ecosystems in western North America. The highest overall predicted impacts as a percent of the ecosystem type are to boreal forest (23-32 percent), shrublands (6-24 percent), and grasslands (9-21 percent). In absolute terms, the largest potential impacts are to shrublands (9.9 to 41.1 million ha). Oil, gas, wind, solar, and geothermal development each have their greatest potential impacts on shrublands. The impacts to shrublands occur in all ecological regions across western North America, but potential impacts are greatest in the North American Deserts (up to 27 percent or 25.8 million ha), Great Plains (up to 24 percent or 8.9 million ha), and Northern Forests (up to 47 percent or 4.3 million ha). Conventional oil and gas development accounts for the largest proportion of the potential impact in all three of these regions. Some states or provinces may experience particularly large impacts to shrublands, including Alberta and Wyoming, where potential for oil and gas development is especially high, and New Mexico, where solar development could potentially affect large areas of shrubland. Understanding the scale of anticipated impacts to these ecosystems through this type of coarse-scale analysis may help to catalyze policy makers to engage in proactive planning.

In Monaco, T.A. et al. comps. 2011. *Proceedings – Threats to Shrubland Ecosystem Integrity*; 2010 May 18-20; Logan, UT. *Natural Resources and Environmental Issues*, Volume XVII. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan Utah, USA.

INTRODUCTION

World demand for energy is projected to increase by 50 percent between 2007 and 2030 (International Energy Agency 2007). This impending rapid development of energy resources may have dramatic consequences for terrestrial ecosystems and wildlife of western North America, because this region is rich in hydrocarbons and has high potential for renewable energy production. Hydrocarbons will remain the largest source of energy worldwide with oil, natural gas, and coal meeting 85 percent of this demand (International Energy Agency 2007). Increasing political uncertainty in many oil-producing nations has prompted accelerating exploitation of North American energy resources, and growing recognition of the potential social and biological ramifications of climate change is driving trends toward increasing development of reduced carbon or carbon-neutral energy sources such as solar, wind, nuclear, and geothermal power (Brooke 2008). The increasing demand for energy and the West's abundant supply

nearly ensures these resources will be developed. If development continues at its current pace, the outcome will likely be "energy sprawl" (McDonald and others 2009), resulting in a western landscape increasingly fragmented by energy infrastructure such as roads, well pads, wind towers, and transmission lines.

Despite growing concerns regarding environmental impacts of energy sprawl, until recently the scope of the cumulative impacts on ecosystems was largely unknown. A recent study measured the potential impacts of major energy sources on terrestrial ecosystems in western North America (Copeland and others in press). Here we summarize the results of Copeland and others (in press) and describe the potential impacts on shrublands, the ecosystem projected to experience the greatest absolute impacts from potential energy development. We describe the energy resources impacting shrublands and the ecological divisions and states or provinces in which shrublands may experience the greatest impacts.

METHODS

We measured potential terrestrial impacts of major hydrocarbon and renewable energy sources across North America (figure 1), including oil and gas, oil shale, oil sands, coal, wind, solar, geothermal, and nuclear (measured as uranium). We did not consider hydropower or biofuels, as those impacts are largely aquatic or the terrestrial impacts have already occurred. More details about the geography and production efficiency for each of these major energy sources can be found in Copeland and others (2011).

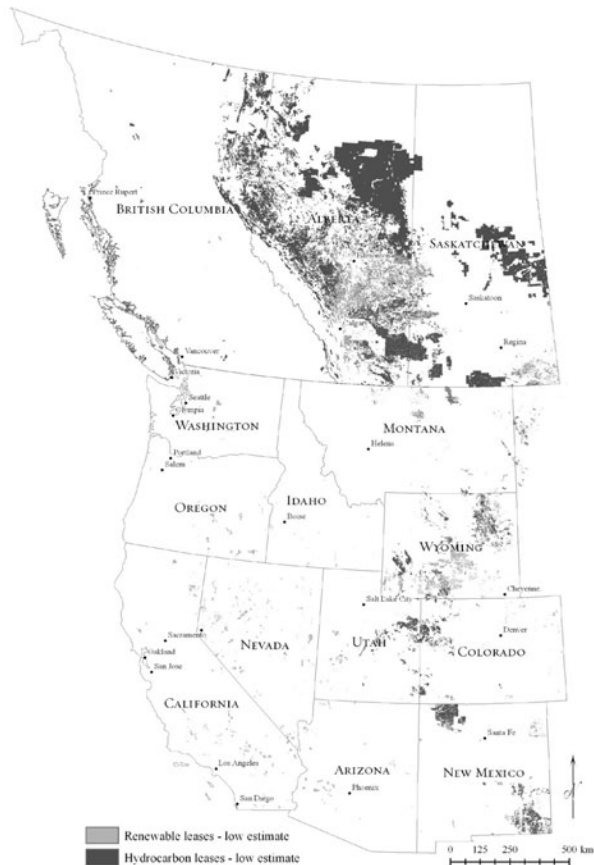


Figure 1. The distribution of leases for renewable and hydrocarbon energy resources across the western North America study area. Renewable leases are displayed over top of hydrocarbon leases, so not all hydrocarbon leases may be shown.

We measured current and potential energy impacts using July 2009 lease and license data from the U.S. National Integrated Lands System database (<http://www.geocommunicator.gov>), Saskatchewan Mineral Disposition Maps and Databases, Alberta Energy, and British Columbia Ministry of Energy and Mines.

For conventional oil and gas development, we determined a low estimate of impact using only leases with currently producing oil and gas wells (IHS incorporated, www.ihsenergy.com) and a high estimate that included all leases. Wind lease data for Canada were unavailable, so we used existing projects to estimate minimum or low impacts (Ventyx Energy 2009). Each wind project point was expanded to represent the land area impacted based on the power production of the project, assuming an impact of 20 ha per MW (US Department of Energy 2008a). We also calculated the footprint of proposed renewable energy zones for wind and solar energy development (Western Governors' Association 2009) to provide a high estimate of the amount of land that may be affected. Although development would not be restricted to these zones, nor would development likely impact the zones entirely, the zones do provide a coarse-scale estimate of the amount of land area that could be affected. Lease data provide an estimate of landscape-scale impacts, including direct and indirect potential future impacts. These datasets were limited to public lands or public subsurface minerals holdings, with the exception of the high estimate for wind and solar development, which incorporated private lands.

We estimated the footprint of energy development on each of five terrestrial ecosystem types: temperate forests, boreal forests, shrublands, grasslands, and wetlands (MEDIAS-France/Postel 2004; ESRI 2006). For shrublands (figure 2), we measured the potential impact of each type of energy development and the amount of shrubland impacted within each ecological division (figure 3, Commission for Environmental Cooperation 1997) and state or province.

RESULTS AND DISCUSSION

Existing and potential energy development could affect, either directly or indirectly, up to 21 percent (96 million hectares) of the five major ecosystems in western North America (Copeland and others 2011). The highest overall predicted impacts as a percent of the ecosystem type are to boreal forest, shrublands, and grasslands (figure 4). In absolute terms, the largest potential impacts are to shrublands; 9.9 to 41.1 million of 169.3 million total hectares may be affected. Oil, gas, wind, solar, and geothermal development each have their greatest potential impacts on shrublands (Copeland and others 2011).



Figure 2. The distribution of shrublands across the western North America study area.

For shrublands, conventional oil and gas development has the greatest current or potential impacts (figure 5). Wind and solar development have the next highest potential impacts on shrublands, but the magnitude of these impacts has greater uncertainty (figure 5). Development of wind and solar resources are expected to rapidly increase, yet face limitations related to electrical transmission and cost. United States and Canadian projections suggest that wind resources may be able to provide for 20 percent of annual electrical energy demand within the next 20 years. This would mean increasing from a current installed capacity of 9669 MW to 348,000 MW, a 36-fold increase (US Department of Energy 2008a; American Wind Energy Association 2009; Canadian Wind Energy Association 2009). Generation of power from solar-photovoltaic and solar-thermal technologies more than doubled in the US between 2000 and 2007, with current capacity at 983 MW. For solar technologies to become more cost effective, 86,000 to 125,000 additional MW need to be installed across the US by 2030 (US Department of Energy 2008b).

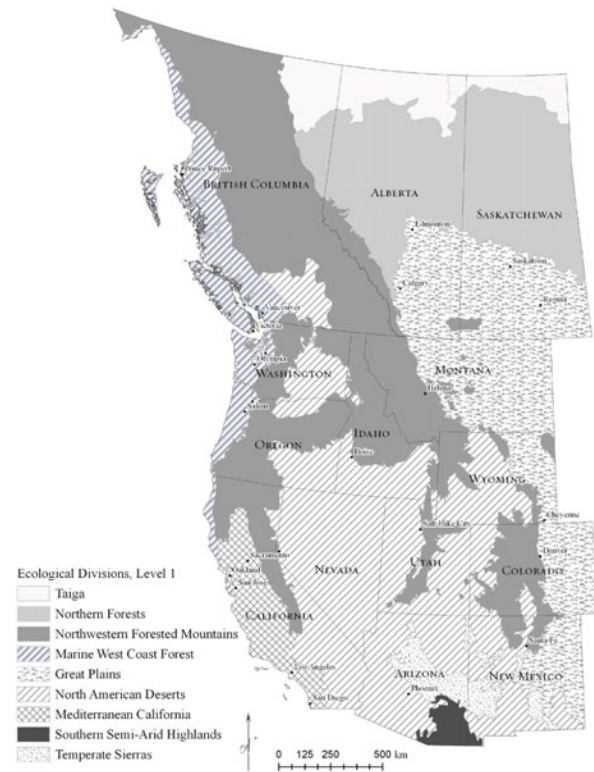


Figure 3. The distribution of level 1 ecological divisions across the western North America study area.

Shrublands are or will be impacted by energy development in all ecological divisions across western North America, but potential impacts are greatest in the North American Deserts, Great Plains, and Northern Forests (figure 6). In the North American Deserts, most energy-related impacts to shrublands would be from conventional oil and gas (2.1 to 7.9 million ha), wind (1.2 to 3.3 million ha) and solar development (60,000 ha to 15.4 million ha). Shrublands in the Great Plains could be most impacted by oil and gas development (2.4 to 5.6 million ha), followed by wind development (65,000 ha to 2.9 million ha) and coal mining (375,000 ha). In the Northern Forests, hydrocarbon extraction could have the greatest potential impacts on shrublands, with most impacts related to conventional oil and gas development (1.3 to 2.9 million ha), followed by oil sands development (859,230 ha) and coal mining (296,000 ha).

Some states or provinces may experience particularly large impacts to shrublands, including Alberta, Wyoming, New Mexico, and Saskatchewan (figure 7). Alberta's shrublands are at the greatest risk of loss or

fragmentation from energy development; 36 percent to 56 percent of Alberta's shrublands could be impacted (figure 7). Most of this impact (65-78 percent) would be from oil and gas development (2.1 to 4 million ha), and oil sands development could also have considerable impacts (891,000 ha). In Wyoming, 15 percent to 42 percent of shrublands could be affected by energy development (figure 7). Oil and gas development also explains most of the potential impact (59-75 percent) in Wyoming (1.3 to 4.6 million ha), and wind development could also impact large areas of Wyoming shrublands (645,000 ha to 1.9 million ha). Shrublands in Saskatchewan are most affected by oil and gas development and coal mining. In New Mexico, Nevada and Utah, most low-estimate energy impacts to shrublands are from oil and gas development, but additional high-estimate impacts are primarily related to solar development.

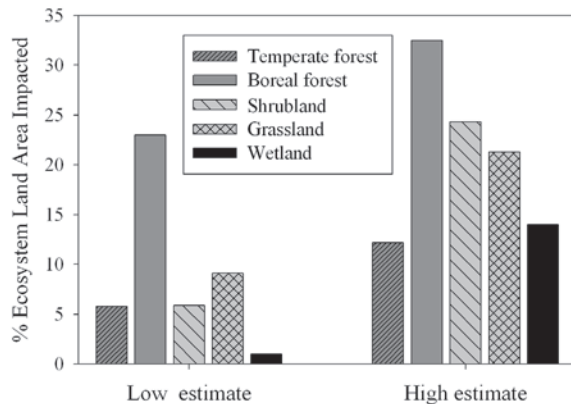


Figure 4. Low and high estimates of the percent of each major ecosystem in western North America that may be impacted by energy development.

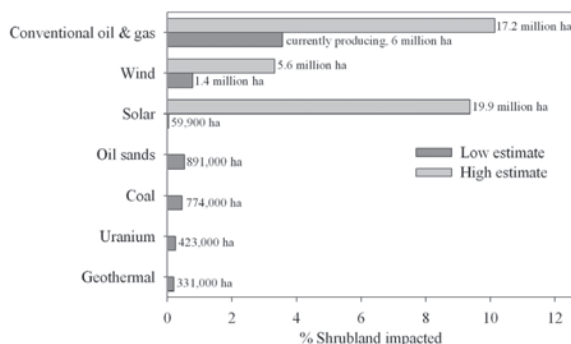


Figure 5. Low and high estimates of the proportion of shrubland ecosystems in western North America that may be impacted by each of seven types of energy development, followed by the numbers of hectares of shrublands that may be impacted.

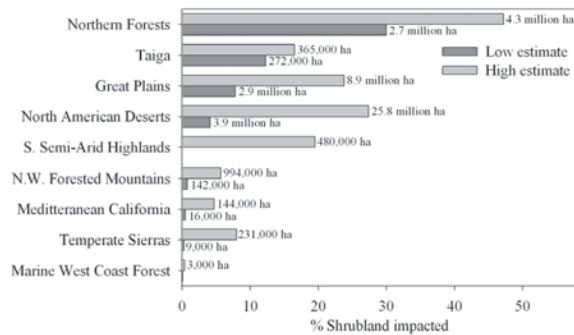


Figure 6. Low and high estimates of the proportion of shrubland ecosystems in each ecological division of western North America that may be impacted by energy development, followed by the numbers of hectares that may be impacted.

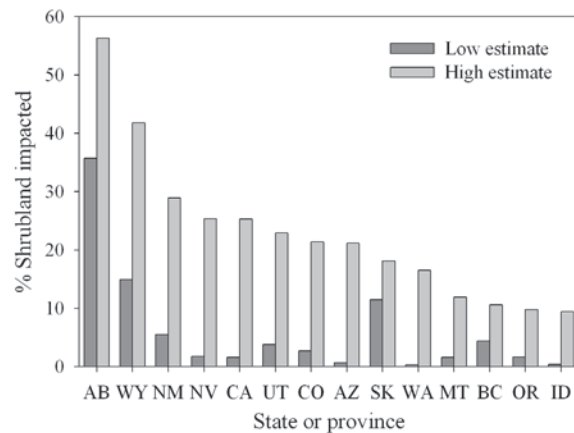


Figure 7. Low and high estimates of the proportion of shrubland ecosystems that may be impacted by energy development in each state or province of western North America: Alberta (AB), Wyoming (WY), New Mexico (NM), Nevada (NV), California (CA), Utah (UT), Colorado (CO), Arizona (AZ), Saskatchewan (SK), Washington (WA), Montana (MT), British Columbia (BC), Oregon (OR), and Idaho (ID).

These potential changes to shrubland ecosystems are alarming, especially because of the limited legal protection these systems currently receive, despite comprising ~30 percent of the land area of western North America and supporting wildlife species such as the greater sage-grouse (*Centrocercus urophasianus*), pygmy rabbit (*Brachylagus idahoensis*), and Wyoming pocket gopher (*Thomomys clusius*) that have recently been considered for protection under the Endangered Species Act. In addition to impacts associated with energy development, shrubland ecosystems and their inhabitants are also suffering under additional

stresses from residential development, invasive species, disease, and climate change. Understanding the scale of anticipated impacts to shrubland and other ecosystems through this type of coarse-scale analysis that highlights ecological and political regions of concern may help to catalyze policy makers to engage in proactive planning, ideally before projects begin, about how to avoid siting conflicts, maintain biodiversity, and determine suitable mitigation responses.

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The Impacts of Fire on Sage-grouse Habitat and Diet Resources

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ABSTRACT

*Small (<40.5-ha) patch fires or mechanical manipulations to reduce big sagebrush (*Artemisia tridentata*) cover has been suggested as a management option to improve sage-grouse pre-nesting and brood rearing habitat and provide a diverse habitat mosaic. We evaluated the effects of prescribed fire and wildfire on sage-grouse habitat in three Wyoming big sagebrush associations (Bluebunch, Thurber's needlegrass, Thurber's needlegrass-Idaho fescue). Response variables included vegetation cover, herbaceous productivity, yield and nutritional quality of forbs preferred by greater sage-grouse, and abundance of common arthropod orders. Wildfire eliminated all sagebrush and >90 percent of the perennial grasses on the Thurber's association. On the Bluebunch association wildfire eliminated sagebrush, but most perennial grasses survived. The prescribed fire on the Thurber's needlegrass-Idaho fescue association removed 95 percent of the sagebrush with most perennial grasses surviving. Habitat cover (shrubs and tall herbaceous cover (> 18cm height)) was 33-90 percent lower after burning compared to unburned controls. The removal of big sagebrush decreased structural cover and reduced or eliminated forage provided by big sagebrush for sage-grouse. This would be potentially damaging in sage-grouse year-round and wintering habitat. Burning reduced Wyoming big sagebrush forage production by about 450 kg/ha on the Thurber's needlegrass-Idaho fescue association. Yields or cover of perennial forbs used by sage-grouse in their diets did not differ between burned sites and not burned sites in the Bluebunch and Thurber's needlegrass-Idaho fescue associations. In the Thurber's needlegrass association long leaf phlox was the only perennial forb to increase after fire. Pale alyssum, a non-native forb, was the dominant annual after fire in the Thurber's needlegrass and Thurber's needlegrass-Idaho fescue associations. Yields or cover of annual forbs used by sage-grouse in their diets increased temporarily after fire in the Bluebunch association. Although cheatgrass increased in the Thurber's association it has remained a minor component of the post-fire community. The abundance of ants (Hymenoptera) decreased after fire while the abundance of other arthropods remained unaffected in the Thurber's needlegrass-Idaho fescue association. The results indicate that prescribed fire will not improve habitat characteristics for sage-grouse in Wyoming big sagebrush steppe where the community already consists of shrubs, native grasses, and native forbs. Burning of Wyoming big sagebrush communities to enhance other species habitat requirements should minimize mortality of native perennial grasses and forb species, result in a mosaic pattern of burned and unburned patches, and avoid areas of critical habitat.*

In Monaco, T.A. et al. comps. 2011. Proceedings – Threats to Shrubland Ecosystem Integrity; 2010 May 18-20; Logan, UT. Natural Resources and Environmental Issues, Volume XVII. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan Utah, USA.

INTRODUCTION

Big sagebrush (*Artemisia tridentata*) steppe is one of the major vegetation types of the western United States and estimates of its historic coverage exceed 600,000 km² (Anderson and others 1998, West 1983; West and Young 2000). Big sagebrush steppe has been fragmented and reduced in area the past 150 years as a result of altered fire regimes, invasive weed dominance, agricultural land conversion, non-native grass seeding, sagebrush removal programs, piñon-juniper (*Pinus-Juniperus*) woodland expansion, and urban and industrial development (Knick and others 2003; Miller and others 2005; Rowland and Wisdom 2005; West 1983; West and Young 2000).

Big sagebrush steppe is delineated into three complexes: Wyoming big sagebrush (*Artemisia tridentata* spp. *wyomingensis*); basin big sagebrush (*Artemisia tridentata* spp. *tridentata* Nutt.); and mountain big sagebrush (*Artemisia tridentata* spp. *vaseyana*). The Wyoming big sagebrush alliance is considered the most vulnerable as it is susceptible to replacement by invasive annual grasses, particularly after fire disturbance (Miller and Eddleman 2001; Suring and others 2005). Large areas of the alliance are in low seral condition or have converted to cheatgrass (*Bromus tectorum*) grasslands. The dominance of cheatgrass has resulted in dramatic increases in both size and frequency of wildfires in the

Great Basin, Snake River Basin, and Columbia Plateau. It is estimated that mean fire return intervals (MFRI) in many Wyoming big sagebrush plants communities have been reduced from 50-100 years to less than 10 years as a result of cheatgrass (*Bromus tectorum*) invasion (Baker 2006; Whisenant 1990; Wright and others 1979). The conversion from native to annual grass-dominated communities has resulted in the loss of habitat and reduced populations of sagebrush obligate and facultative wildlife species (Connelly and Braun 1997; Connelly and others 2000a; Crawford and others 2004; Welch 2002).

Southeastern Oregon, northern Nevada, and southwestern Idaho retain extensive Wyoming big sagebrush communities in mid to late seral ecological stages. These areas are co-dominated by big sagebrush and perennial bunchgrasses with little cheatgrass present (Davies and others 2006). While large scale application of fire is not recommended for this alliance, the use of small (<40.5-ha) patch fires and mechanical manipulations to reduce big sagebrush cover has been recommended as a management option to improve sage-grouse pre-nesting and brood-rearing habitat and provide a diverse habitat mosaic for other species (Connelly and others 2000b; Dahlgren and others 2006; Hagen 2005; Helmstrom et al. 2002; Petersen and Best 1987). Thinning dense stands of sagebrush or creating open patches of herbaceous vegetation has been suggested as methods to increase herbaceous cover and forb production (Dahlgren and others 2006; Wirth and Pyke 2003). Forbs amount to 50-80 percent of the diet of sage-grouse during pre-nesting and brood-rearing periods in the spring (Barnett and Crawford 1994; Drut and others 1994).

The effect of prescribed fire on habitat characteristics of big sagebrush steppe, specific to sage-grouse, has produced both variable and consistent results. Whether fires are large, small, or mosaic in pattern, the loss of big sagebrush reduces structural cover for successful nesting and concealment, as well as decreasing forage provided by sagebrush (Crawford and others 2004; Davies and others 2007). In Wyoming big sagebrush communities, burning has not been effective at increasing total forb diversity or abundance, although productivity of individual forb species has increased (Bates and others 2009; Beck and others 2009; Fischer and others 1996; Nelle and others 2000; Wroblewski and Kauffman 2003). Insects are an important dietary component of young sage-

grouse and may comprise 75-100 percent of the diet the first several weeks after hatching (Gregg 2006; Johnson and Boyce 1990; Patterson 1952). Fire in a Wyoming big sagebrush community in Idaho reduced the abundance of ants while beetles, crickets, and grasshoppers were unaffected (Fischer and others 1996; Rickard 1970).

We evaluated the effects of wildfire and prescribed fire on sage-grouse habitat characteristics in three Wyoming big sagebrush plant associations by measuring impacts to; 1) vegetation cover requirements developed by (Connelly and others 2000b) for sage-grouse habitat, 2) the productivity of forb species utilized by sage-grouse in their diets, and 3) the abundance of arthropods.

STUDY AREAS

We conducted post-fire studies on the Northern Great Basin Experimental Range (NGBER), 56 km west of Burns, Oregon, and the Sheepshead Mountains, 110 km southeast of Burns. Three Wyoming big sagebrush associations were evaluated. The sites were representative of mid to high seral Wyoming big sagebrush plant communities with a mix of big sagebrush, native grasses, and native forbs. Big sagebrush and total herbaceous cover values were representative of Wyoming big sagebrush associations in eastern Oregon (Davies and others 2006). Sites were located in year-round sage-grouse habitat and were within 1-5 km of active leks. Vegetation cover values met sage-grouse nesting and brood-rearing habitat requirements for arid big sagebrush sites as suggested by Connelly and others (2000b). Climate is typical of the northern Great Basin with most precipitation arriving in winter and early spring, whereas summers are warm and dry. Annual precipitation at Burns, Oregon, has averaged about 280 mm since the 1930s. Drought occurred in 2000-2002 and 2007 and precipitation was below average in 2003, 2004, and 2008. Precipitation was above average in 2005, 2006, and 2009. General references used for plant identification were Hitchcock and Cronquist (1987) and the Natural Resource Conservation Service (2009).

The site at the NGBER was a Wyoming big sagebrush/Thurber's needlegrass (*Achnatherum thurberianum*)-Idaho fescue (*Festuca idahoensis*) association (Thurber's-Idaho fescue Association). Elevation is 1400 m and topography is flat (0-2

percent slope). Soils are a complex of four series sharing several attributes; all are Durixerolls, soil surface texture is sandy loam to loamy sand, and are well drained with a duripan beginning between 40-75 cm deep (Lentz and Simonson 1986). Wyoming big sagebrush was the dominant shrub and basin big sagebrush and green rabbitbrush (*Chrysothamnus viscidiflorus*) were subdominant shrubs. Idaho fescue and Thurber's needlegrass were the main perennial bunchgrasses. Sandberg's bluegrass (*Poa secunda*), bluebunch wheatgrass (*Pseudoroegneria spicata*), prairie Junegrass (*Koeleria macrantha*), and bottlebrush squirreltail (*Elymus elymoides*) were subdominant perennial grasses. Common perennial forbs were of taper-tip hawksbeard (*Crepis acuminata*), milkvetch (*Astragalus* spp.), fleabanes, (*Erigeron* spp.), western groundsel (*Senecio intergerrimus*), common yarrow (*Achillea millefolium*), and long-leaved phlox (*Phlox longifolia*). Common annual forbs were little blue-eyed Mary (*Collinsia parviflora*), slender phlox (*Microsteris gracilis*), and non-native pale alyssum (*Alyssum alyssoides*). Wyoming big sagebrush cover averaged 10 percent (range 6-17 percent) and grass-forb cover exceeded 15 percent (Davies and others 2007). Cheatgrass was present in trace amounts (<1 percent cover). Prior to livestock removal in 1999, grazing by cattle was of moderate use (40-50 percent utilization), using a rest rotation system.

Sites on the Sheephead Mountains included two plant associations: Wyoming big sagebrush/bluebunch wheatgrass (Bluebunch) and Wyoming big sagebrush/Thurber's needlegrass (Thurber's). Light cattle grazing occurred in the winter and early spring. The Thurber's association sites were in mid seral and high seral condition. Elevation was about 1280 m and sites were located on lake terraces (0-5 percent slope) created during the Pleistocene epoch. Soils were a complex of Durixerolls, Haplocambids, and Haploargids. Wyoming big sagebrush was the dominant shrub and green rabbitbrush, spiny hopsage (*Atriplex spinosa*) and gray horsebrush (*Tetradymia canescens*) were present in low densities. Thurber's needlegrass was the main perennial bunchgrasses. Sandberg's bluegrass and bottlebrush squirreltail were subdominant grasses. Common perennial forbs were taper-tip hawksbeard, Nevada lomatium (*Lomatium nevadense*), lava aster (*Aster scopulorum*), fleabane species, and long-leaved phlox. Annual forbs were represented by little blue-eyed Mary and pale alyssum. Cheatgrass was present in trace amounts (<1 percent cover).

Bluebunch association sites were rated in high seral condition. Sites were located on hillslopes and ridges with slopes between 5-40 percent at elevations between 1300-1480 m. Soils included Argixerolls, Paleargids, and Haploargids. Wyoming big sagebrush was the dominant shrub and green rabbitbrush was present in low densities. Bluebunch wheatgrass was the main perennial bunchgrasses with Sandberg's bluegrass and Cusick's bluegrass (*Poa cusickii*) as subdominant perennial grasses. Common perennial forbs consisted of western hawksbeard (*Crepis occidentalis*), prairie lupine (*Lupinus lepidus*), milkvetch, low pussytoes (*Antennaria dimorpha*), taper-tip onion (*Allium acuminata*), Lomatium spp. (*Lomatium* spp.), lava aster, fleabanes, and Hoods phlox (*Phlox hoodii*). Annual forbs were represented by little blue-eyed Mary, narrow-leaf collomia (*Collomia linearis*), slender phlox, willow-weed (*Eppilobium paniculatum*), and rocketstar (*Lithophragma bulbifera*). Cheatgrass was present in trace amounts.

METHODS

NGBER

We used a randomized complete block design to compare vegetation response variables and arthropod abundance between burned (burn) and not burned (control) for the Thurber's-Idaho fescue association. We established five 4-ha blocks in 2001 and within each block were two 2-ha plots, with one plot randomly assigned to be burned. Prescribed burning was done in late September and early October 2002. The burn application was a strip head fire, ignited using a gel-fuel terra torch (Firecon, Inc., Ontario, Oregon). Wind speeds were between 5-20 km/hr, air temperatures were 20°-25° C, and relative humidity varied from 10-35 percent during prescribed burns. Moisture content of fine fuels (herbaceous vegetation) was 8-12 percent and fine fuel loads were 350-420 kg/ha. Burns were complete across treatment plots and killed 92 percent of Wyoming big sagebrush. We randomly placed six 50-m transects within each treatment plot in 2001. Transects were permanently marked using 40 cm rebar stakes. We measured plant species cover in June 2001-2006, 2008, and 2009. Shrub canopy cover was measured by species using the line intercept technique and excluded canopy gaps >15 cm from measurements (Canfield 1941; Boyd and others 2007).

Herbaceous canopy cover (by species), bare ground and rock, litter, and biological crust (moss, lichen, algae) was estimated inside 40 × 50-cm frames (0.2 m²) located at 3-m intervals on each transect line. Herbaceous yield was gathered by clipping at the functional group level (Sandberg's bluegrass, perennial bunchgrasses, perennial forbs, annual forbs, and cheatgrass) in mid-June 2002-2008. From 2004-2009, we measured forb yield by species in mid-April, mid-May, and mid-June. We collected data at these periods to determine availability of dietary forbs used by sage-grouse from late breeding through brood rearing periods. Forb species that are consumed by sage-grouse were determined from review of the literature (Barnett and Crawford 1994; Drut and others 1994; Klebenow and Gray 1968; Nelle and others 2000; Peterson 1970; Wallestad and others 1975). Perennial grasses were clipped to a 2-cm stubble height. Cheatgrass and forbs (perennial and annual) were clipped to ground level. Perennial grasses and forbs were clipped in 15, 1-m² randomly located frames per 2-ha plot each sampling period. Annual forbs and cheatgrass were clipped in 0.20-m² nested plots inside 1-m² frames. Clipped samples were oven dried at 56° C for 48 hours. Perennial and annual forbs were weighed by species or tribes. Arthropods were collected using pitfall traps containing a 1: 4 mixture of antifreeze and water. In each plot we randomly placed 10 traps each collection period. Traps were sampled once a week during 2-week periods in early May and early June of 2004-2005. Captured arthropods were identified to Order and counted.

We used a repeated measures analysis of variance (ANOVA) PROC MIXED procedure for a randomized complete block design to compare time, treatment, and time by treatment interactions for plant species cover, forb and grass yield, and arthropod counts (SAS Institute, Cary, North Carolina). We evaluated vegetation canopy cover by grouping species according to sage-grouse habitat guidelines: big sagebrush, green rabbitbrush, total herbaceous, tall herbaceous (>18 cm ht), perennial grasses (>18 cm ht), perennial forbs, and annual forbs (Connelly et al. 2000b). We categorized yield by life form: perennial grass, Sandberg's bluegrass, perennial forb, annual forb, sage-grouse dietary perennial and annual forbs, pale alyssum, and total herbaceous biomass. We tested data for normality using the univariate procedure and arcsine-square root transformed data when normality failed to stabilize variance. We report back transformed means and set statistical significance of all tests at $P < 0.05$.

Sheepshead Mountains

Initial vegetation measurements were made in June 2001, as part of a study assessing plant cover potentials in Wyoming big sagebrush, on 15 sites. Twelve of the plots burned in a wildfire in August 2001 that encompassed 16,000 ha. Unburned patches remaining within the fire perimeter and the fires perimeter were used to serve as unburned controls. The Thurber's association had three control sites (1 mid-seral, 2 high seral) and four burned sites (2 mid-seral, 2 high seral). The Bluebunch association had seven high seral control sites and seven high seral burned sites. Control plant association sites were located within 0.1-5 km of burned sites. Vegetation cover values and composition of control sites were not different when compared to preburn values of the burned sites. Five 50-meter transects were permanently established on each site. Shrub canopy cover (by species) was determined using the line-intercept method as described above. Herbaceous species canopy cover and density, bare ground and rock, litter, and biological crust were estimated using 0.2 m² frames. Frames were located every 3 m on transect lines (15 frames per transect/ 75 frames per plot). A completed randomized repeated measures generalized linear model (GLM) PROC MIXED compared year, treatment (burn, control), and year by treatment interaction between burned and control sagebrush steppe for plant vegetation covers and density. Associations were analyzed separately. We evaluated vegetation canopy cover by grouping species according to sage-grouse habitat guidelines as described above. Mean separations, transforms, and significance level were performed as above.

RESULTS

NGBER Prescribed Fire

The prescribed fire significantly altered cover and biomass values in the burn treatment compared to the control. The dynamics of herbaceous cover and biomass were in response to year x treatment interactions resulting from initial fire effects (2003) and post-fire weather. In the first year after fire (2003), cover of tall herbaceous and perennial grasses was 25 percent lower in the burn than the control (2003) (table 1). After 2003, there were no treatment differences for tall herbaceous ($P = 0.542$) and perennial grass cover ($P = 0.458$). Tall herbaceous cover was primarily composed of perennial grasses as tall forb cover did not exceed 1 percent in either

treatment. In both treatments, perennial grass cover increased 20 percent between 2004 and 2006 in response to favorable growing conditions and then declined in 2008 as a result of below average precipitation ($P < 0.001$). Cover of perennial forb species and tall forbs (≥ 18 cm) did not differ between the burn and control ($P = 0.547$) or across years ($P = 0.804$). Annual forb cover was greater in the burn than the control in most years after fire ($P = 0.012$; table 1). Nearly all annual forb cover in the burn consisted of pale alyssum, an introduced old world weed. Cover of other annual forbs did not increase after fire and there were no differences compared to the control ($P = 0.689$). Year x treatment interactions

were significant for total herbaceous, perennial grass (≥ 18 cm height), tall herbaceous (≥ 18 cm height), big sagebrush, green rabbitbrush, and annual forb cover. In 2005, 2006, 2008, and 2009 total herbaceous cover was 21-35 percent greater in the burn than the control ($P < 0.001$). Wyoming big sagebrush cover was lower in the burn after fire and in 2009 was about 10 percent of pre-burn cover ($P < 0.001$; table 1). Sagebrush cover in the burn was provided by surviving plants as there was no recruitment of new individuals. Green rabbitbrush cover was reduced the first year ($P < 0.001$) after fire; recovering to pre-burn levels after 2004 ($P = 0.085$; table 1).

Table 1. Prescribed fire effects on herbaceous cover (%) in the Wyoming big sagebrush/Thurber's needlegrass-Idaho fescue association, Northern Great Basin Experimental Range, Oregon. Response variable data are means and standard errors for preburn (2002), first year post-fire (2003), and post-fire average (2004-2009). Data for 2004 -2009 present the range of values for this time period. Italicized values and different lower case letters indicate significant differences between treatments.

Response variable	Preburn		Post burn - 2003		Post-burn 2004-2009	
	Burn (%)	Control (%)	Burn (%)	Control (%)	Burn (%)	Control (%)
Pere. Grass ^{1,2} Range ³	15.2 $\pm$ 0.9	14.5 $\pm$ 1.1	9.4 $\pm$ 0.6 a	13.2 $\pm$ 1.2 b	15.8 $\pm$ 1.8 (13 - 22.7)	14.5 $\pm$ 1.8 (10.2 - 20.9)
Cheatgrass Range	0.1 $\pm$ 0.0	0.1 $\pm$ 0.0	0 $\pm$ 0	0 $\pm$ 0	0.6 $\pm$ 0.3 (0 - 1.2)	0.3 $\pm$ 0.2 (0 - 0.8)
Perennial Forb ⁴ Range	0.3 $\pm$ 0.1	0.5 $\pm$ 0.2	0.2 $\pm$ 0.1	0.3 $\pm$ 0.1	0.5 $\pm$ 0.1 (0.2 - 0.6)	0.7 $\pm$ 0.2 (0.4 - 1.0)
Annual Forb Range	0.5 $\pm$ 0.2	0.3 $\pm$ 0.1	1.2 $\pm$ 0.4 b	0.35 $\pm$ 0.1 b	5.2 $\pm$ 1.9 b (2.0 - 12.5)	0.7 $\pm$ 0.4 a (0.4 - 1.1)
Dietary Ann. Forb; Range	0.2 $\pm$ 0.1	0.2 $\pm$ 0.1	0.6 $\pm$ 0.2 b	0.2 $\pm$ 0.1 a	0.3 $\pm$ 0.1 (0.2 - 0.6)	0.2 $\pm$ 0.1 (0.1 - 0.4)
Herbaceous Range	15.9 $\pm$ 1.4	15.3 $\pm$ 1.7	10.7 $\pm$.6 a	13.8 $\pm$ 1.3 b	21.3 $\pm$ 2.5 b (15 - 27.6)	15.9 $\pm$ 1.9 a (11.5 - 22.6)
Sagebrush Range	10.2 $\pm$ 1.3	10.8 $\pm$ 1.1	0.3 $\pm$ 0.1 a	10.6 $\pm$ 1.1 b	0.7 $\pm$ 0.1 a (0.5 - 0.9)	10.5 $\pm$ 0.4 b (9.2 - 11.3)
Rabbitbrush Range	2.7 $\pm$ 0.9	3.0 $\pm$ 0.7	0.6 $\pm$ 0.4 a	3.0 $\pm$ 0.7 b	4.1 $\pm$ 0.6 (2 - 5.4)	3.1 $\pm$ 0.3 (2.4 - 4.0)
Habitat Cover Range	28.9 $\pm$ 1.8	29.1 $\pm$ 1.8	10.5 $\pm$ 0.8 a	27.0 $\pm$ 0.5 b	21.0 $\pm$ 2.2 a (16.5 - 29.4)	28.8 $\pm$ 2.3 b (22.7 - 36.5)

¹Perennial bunchgrasses and Sandberg's bluegrass.

²Perennial grass cover represented 95% of tall herbaceous cover which are plants typically > 18 cm tall.

³Cover range across years (2004-2009).

⁴All perennial forbs measured for cover and yield were forbs that are consumed by sage-grouse.

Table 2. Prescribed fire effects (post-burn) to herbaceous cover (%) and yield (kg/ha) in the Wyoming big sagebrush/Thurber's needlegrass-Idaho fescue associations, Northern Great Basin Experimental Range, Oregon. Response variable data are pre-burn, first year post-burn (2003), and 6-year post-fire average (2004-2009). Data for 2004 -2009 also present range of values for this time period. Italicized values and different lower case letters indicate significant differences between treatments for yield within respective time periods.

Response variable	Preburn		Post burn - 2003		Post-burn 2004-2009 average	
	Burn (kg/ha)	Control (kg/ha)	Burn (kg/ha)	Control (kg/ha)	Burn (kg/ha)	Control (kg/ha)
Perennial Grass ¹ Range ²	192 ± 8	207 ± 21	166 ± 10 <i>a</i>	211 ± 28 <i>b</i>	449 ± 75 <i>b</i> (287-910)	235 ± 48 <i>a</i> (146-449)
Poa secunda Range	44 ± 5	59 ± 16	11 ± 1 <i>a</i>	27 ± 3 <i>b</i>	95 ± 25 <i>b</i> (45-189)	52 ± 11 <i>a</i> (22-79)
Perennial Forb ³ Range	37 ± 4	47 ± 21	7 ± 1 <i>a</i>	30 ± 3 <i>b</i>	49 ± 6 (33-66)	47 ± 6 (31-65)
Annual Forb Range	13 ± 2	10 ± 2	4 ± 0.4	6 ± 2	98 ± 14 <i>b</i> (46-143)	27 ± 3 <i>a</i> (16-38)
Dietary Ann. Forb; Range	2 ± 0.2	2.0 ± 0.1	11 ± 4 <i>b</i>	2 ± 1 <i>a</i>	2 ± 0.4 (0.5-2.5)	1 ± 0.4 (0.2-1.5)
Herbaceous Range	318 ± 14	324 ± 56	187.2 ± 2.2 <i>a</i>	276 ± 32 <i>b</i>	692 ± 95 <i>b</i> (366-1096)	363 ± 59 <i>a</i> (217-613)

¹Large deep-rooted perennial bunchgrasses.

²Yield range across years (2004-2009).

³All perennial forbs measured for cover and yield were forbs that are consumed by sage-grouse.

Herbaceous yield was greater in the burn than the control treatment by the second year after fire ($P < 0.001$) (table 2). Herbaceous, tall herbaceous, and perennial grass yield was about twice as great in the burn than the control from 2005 to 2009 ($P < 0.001$). Perennial forb yield did not increase after the fire and did not differ from the control. Annual forb yield was greater ($P < 0.001$) in the burn throughout the study. Pale alyssum increased in the burn and comprised 90 percent of forb (annual and perennial) yield after fire ($P > 0.001$).

Yields of dietary forb species demonstrated only a few short-term differences between treatments. Yields of *Astragalus* spp. were greater after fire in June sampling periods (2004 to 2009) (burn, 32.3 ± 1.62 kg/ha; control, 21.9 ± 1.0 kg/ha; $P > 0.045$). However, *Astragalus* yields did not differ during April ($P = 0.878$) and May ($P = 0.923$) sampling periods. Other species of perennial forbs utilized by sage-grouse did not differ in yield between treatments. These included yields of the Cichorieae tribe ($P = 0.317$) and long-leaved phlox ($P = 0.76$). The yield of annual forbs that

sage-grouse utilize in their diet was 8-10 times greater in the burn than the control in 2003 and 2004 ($P > 0.001$). On other sample dates and across the study period dietary annual forbs did not differ in yield between the burn and control ($P = 0.126$). Slender phlox and blue-eyed Mary were the dietary annuals collected. The number of ants captured was 135 percent and 175 percent greater in the control than the burn in 2004 and 2005, respectively. Beetle captures did not differ between treatments ($P = 0.504$). The number of grasshoppers and crickets captured was 200 percent higher in the burn ($P = 0.014$) in 2005, while more caterpillars were captured in the control in 2004 (46 percent greater) and 2005 (135 percent greater) ($P = 0.036$).

Sheepshead Wildfire

The Sheepshead wildfire eliminated Wyoming big sagebrush on all burned sites. Sagebrush seedlings began appearing on several sites in 2004, but at very low densities. Green rabbitbrush re-sprouted the first year after fire but its density remained low across the plots throughout the study period (< 20 plants/ha, < 1

percent cover). Spiny hopsage and gray horsebrush present in the Thurber's needlegrass association were eliminated by the fire. Herbaceous response varied by association. Moss and other biological crust were virtually eliminated by fire and remained well below pre-burn levels on both associations eight years after fire.

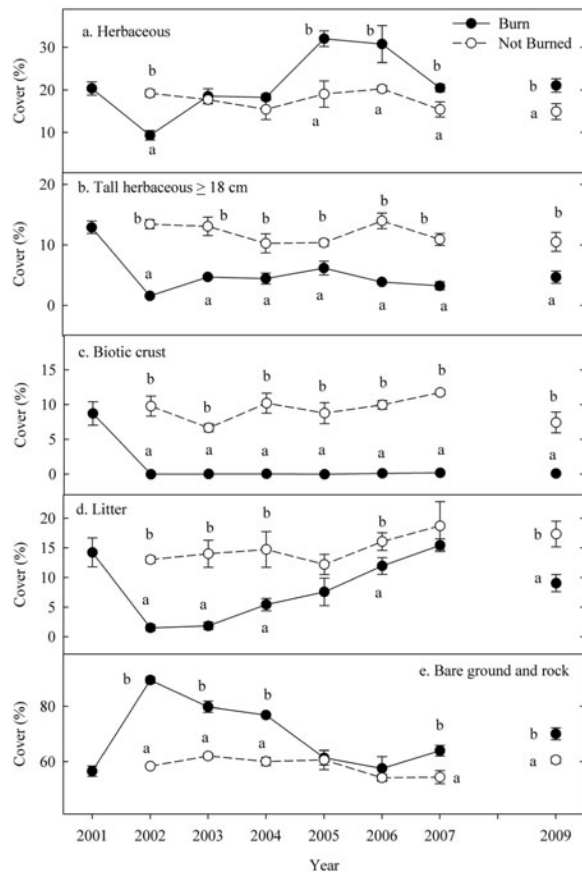


Figure 1. Ground covers (%) for the burned and not burned sites, Wyoming big sagebrush/Thurber's needlegrass association, Sheephead Mountains, Oregon (2001-2009); (a) herbaceous cover, (b) tall herbaceous cover (>18 cm ht), (c) biotic crust, (d) litter cover, and (e) bare ground and rock. Data are in means $\pm$ SE. Means sharing a common lower case letter are not significantly different ($P>0.05$).

Wyoming Big Sagebrush/Thurber's Needlegrass Association

The Wyoming big sagebrush/Thurber's needlegrass association was severely impacted by the wildfire by altering herbaceous composition and reducing habitat cover (table 3). Sagebrush cover was reduced from an average of 17.3 ± 1.5 percent to zero. Herbaceous cover declined the year after fire but exceeded pre-burn and not burned levels from 2003-2009 (figure 1a; $P>0.001$). However, the fire resulted in an average 64

percent reduction (range 40-88 percent) in tall herbaceous cover (>18 cm) between 2003 and 2009 (figure 1b; $P>0.001$). Micro-biotic crust [primarily twisted moss (*Tortula ruralis*)] and litter were mostly located beneath sagebrush canopies prior to the fire. The fire eliminated the micro-biotic component and reduced litter cover by 17-88 percent depending on year (figure 1c and 1d; $P>0.001$). Litter cover in the burn has increased, however, eight years after the fire it was nearly 50 percent lower than not burned sites ($P>0.001$). Bare ground averaged 21 percent greater in the burn (range 31-53 percent) (figure 1e; $P>0.001$).

Herbaceous compositional changes were reflected by reductions of perennial grass cover and increased cheatgrass and annual forb covers. Perennial bunchgrasses and Sandberg's bluegrass have been slow to respond, as the fire killed the majority of these plants (figure 2a and b). Perennial bunchgrass cover has, on average, been 71 percent lower (range 53-93 percent) than non-burned sites. The reduction in grass cover accounted for most of the loss in tall herbaceous cover. Thurber's needlegrass density was reduced 90 percent to less than 1 plant/m² from a pre-burn level of 5.6 ± 0.7 plants/m² ($P>0.001$). In 2009, perennial grass density was 1.6 ± 0.6 plants/m², 75 percent lower than non burned sites. Cover of perennial forbs did not differ between burned and non-burned sites (figure 2c). However, species responses resulted in altered composition for this life-form group. In the burn sites, longleaf phlox increased by an average of 600 percent while other forb species decreased or were unaffected by the fire. Mat forming perennial forb cover was reduced by >90 percent, including oval-leaved buck-wheat (*Eriogonum ovalifolium*) ($P=0.024$), lava aster ($P=0.037$), Hoods phlox ($P=0.008$), and fleabane species ($P=0.047$). Cover of cheatgrass ($P=0.007$) and annual forbs ($P>0.001$) increased after fire (figure 2d and 2e). Cheatgrass cover has averaged about 5 percent cover (20 percent of total herbaceous cover) the past 4 measurement years (2005 to 2009). Annual forb cover was mainly represented by native species the first year after fire, after which annual forbs became increasingly comprised of pale alyssum. Pale alyssum has represented about 50 percent of total herbaceous cover the past 4 measurement years.

Although the results indicate that perennial forbs typically consumed by sage-grouse increased after fire, this change was entirely a result of greater long-

leaf phlox cover (figure 3a; $P>0.001$). Other perennial forbs utilized by sage-grouse either did not increase or declined in cover. The cover of annual forbs consumed by sage-grouse was 5 to 10 times greater the first three years after fire on burn sites ($P>0.001$). Species included willow weed ($P>0.001$), slender phlox ($P=0.042$), little blue-eyed Mary ($P=0.036$), and fireweed (*Gayophytum* spp.; $P>0.001$).

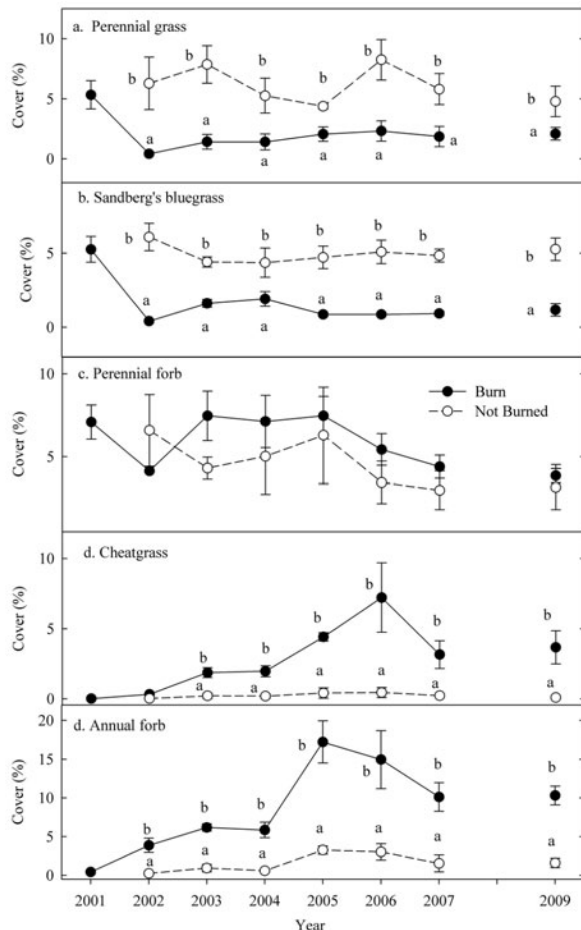


Figure 2. Herbaceous covers (%) for the burned and not burned sites, Wyoming big sagebrush/Thurber's needlegrass association, Sheepshead Mountains, Oregon (2001-2009); (a) perennial grass (e.g. Thurber's needlegrass and other large bunchgrasses), (b) Sandberg's bluegrass, (c) perennial forb, (d) cheatgrass, and (e) annual forb. Data are in means $\pm$ SE. Means sharing a common lower case letter are not significantly different ($P>0.05$).

Wyoming Big Sagebrush/Bluebunch Wheatgrass Association

The wildfire in the Wyoming big sagebrush/bluebunch wheatgrass association moderately impacted herbaceous composition and habitat cover (table 3).

Sagebrush cover was reduced from an average of 12.4 ± 1.2 percent to zero. Total herbaceous and tall (≥ 18 cm) herbaceous cover declined 50 percent the first year after fire, however, since the second year after fire burned sites have not differed from not burned sites (2003-2009) (figure 4a and 4b; $P=0.459$). Micro-biotic crust (primarily twisted moss) was largely eliminated by the fire and has been barely detectable in subsequent years (figure 4c; $P>0.001$). Litter cover decreased 75 percent and remained lower in the burn until 2006, the fifth year after fire (figure 4d). Bare ground averaged 44 percent greater in the burn (range 23-73 percent greater; $P>0.001$) (figure 4e).

Herbaceous compositional changes varied depending on functional group and plant species and have been of short to longer term duration (table 3). Perennial bunchgrass cover was reduced 50 percent the first year after fire; however, after 2003 (second year post-fire) there were no differences between burned and not burned sites (figure 5a). Bluebunch wheatgrass cover was 5-12 percent greater in the burn sites than not burned sites from 2004-2009 ($P>0.007$). Density of bluebunch wheatgrass was unaffected by the fire, averaging 7.8 plants/m² (site range 5-11 plants/m²).

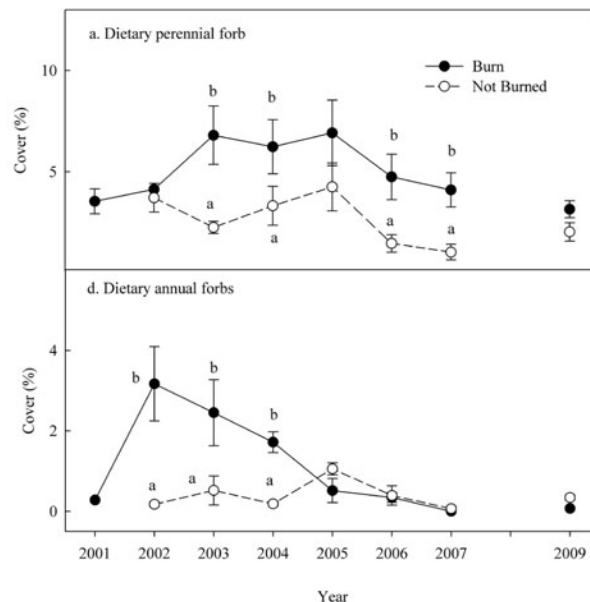


Figure 3. Cover of (a) dietary perennial and (b) dietary annual forbs that are utilized by sage-grouse, burned and not burned sites, Wyoming big sagebrush/Thurber's needlegrass association, Sheepshead Mountains, Oregon (2001-2009). Data are in means $\pm$ SE. Means sharing a common lower case letter are not significantly different ($P>0.05$).

Table 3. Wild and prescribed fire effects to herbaceous species in Wyoming big sagebrush associations, Sheephead Mountains and NGBER, Oregon, 2001-2010. Italicized species are non-native weeds.

Severely Reduced ¹	Slightly Reduced ²	No Effect or Increased ³
<u>Grasses</u> Cusick's bluegrass ^a Indian ricegrass ^a Thurber's needlegrass ^a Sandberg's bluegrass ^a <u>Perennial Forbs</u> desert yellow fleabane ^a dwarf yellow fleabane ^a Hood's phlox ^a Hook's daisy lava aster ^a low pussytoes ^a oval-leaf eriogonum ^a scabland fleabane ^a	<u>Grasses</u> bottlebrush squirreltail ^a <u>Perennial Forbs</u> daggerpod desert yellow fleabane ^b dwarf yellow fleabane ^b Hood's phlox ^b low pussytoes ^b lowly penstemon ^b morning milkvetch obscure milkvetch oval-leaf Eriogonum ^b short-lobed penstemon ^b wooly-pod milkvetch <u>Annual forbs</u> white daisy tidytips ^a	<u>Grasses</u> basin wildrye ^{a,b} bluebunch wheatgrass ^{a,b} bottlebrush squirreltail ^b Cusick's bluegrass ^b Idaho fescue ^b Indian ricegrass ^b junegrass ^{a,b} needle-and-thread ^b Sandberg's bluegrass ^b Thurber's needlegrass ^b <i>cheatgrass</i> ^{a,b} six weeks fescue ^a <u>Perennial Forbs</u> basalt milkvetch big seed lomatium ^a Bolander's yampah broadsheath lomatium ^a Brunea mariposa lily lava aster ^b long-leaf phlox low hawksbeard ^a Nevada lomatium Nevada onion one-stemmed groundsel prairie lupine sickle milkvetch spurred lupine speckle pod milkvetch taper-tip hawksbeard taper-tip onion ^a Tolmie's onion ^a western hawksbeard ^a <u>Annual forbs</u> autumn willow-weed blepharipappus ^a Cyrptantha spp. groundsmoke spp. little blue-eyed Mary pink microsteris pinnate tansy mustard sinuate gilia ^a thread-stem linanthus ^a thread-leaf phacelia ^a white-stemmed blazing star <i>desert alyssum</i> <i>burr buttercup</i> <i>Jim Hill tumble mustard</i> ^a <i>yellow salsify</i>
¹ Severely reduced – species cover reduced more than 80% with no change in years following fire. ² Slightly reduced – species cover between 50% -90% of pre burn levels the first 3 years after fire. ³ No effect or increased – Cover not affected or increased above pre-burn levels within three years after fire. ^a Sheephead wildfire, August 2001. ^b NGBER prescribed fire, Sept-Oct 2002.		

Other perennial bunchgrass species (Idaho fescue, Cusick's bluegrass) were reduced by greater than 95 percent in density ($P>0.001$) and cover ($P>0.001$) and have not recovered. There was a significant reduction in cover of Sandberg bluegrass between 2002 and 2007 (figure 5b). Density of Sandberg bluegrass was reduced by 75 percent after fire, to less than 7 plants/m² ($P=0.008$). Not until 2009, 8 growing seasons after fire, did Sandberg bluegrass cover equalize between burned and not burned sites, though density was about 25 percent lower in the burned areas. Total perennial forb cover did not differ between burned and unburned sites. Perennial forb cover increased 100 percent in burned and not burned sites in response to higher precipitation in 2005, compared to other years. Mat forming perennial forbs were reduced by >95 percent, however, they represented only a small portion (<10 percent) of total perennial forb cover. Cheatgrass cover was 10 times greater in the burned sites in 2009 ($P>0.001$). However, cheatgrass in the burned sites was a small component of the herbaceous layer, representing less than 5 percent of total herbaceous cover.

Annual forb cover fluctuated by year and was 200-400 percent greater in the burned sites from 2003 to 2006 ($P>0.001$). Annual forb cover has not differed between burned and not burned sites the past two measurement years (2007, 2009; $P=0.589$). Annual forbs were dominated by little blue-eyed Mary, representing between 60 to 90 percent of dietary annual forb cover. Other annuals increasing after the fire were willow weed ($P=0.034$), *Cyrptantha* spp. (*cyrptantha*) ($P=0.011$), *Descurainia pinnata* (pinnate-tansy mustard) ($P=0.004$), and a non-native mustard *Sisymbrium altissimum* (Jim Hill tumble-mustard) ($P=0.016$). Perennial forbs utilized by sage-grouse did not increase in cover after fire (fig 6a; $P=0.784$). Covers of annual forbs consumed by sage-grouse were 6 to 10 times greater from 2003 to 2006 in the burn sites (figure 3b; $P>0.001$). Species included willow weed, slender phlox, and little blue-eyed Mary.

DISCUSSION

Cover and Composition

The impact of fire on sage-grouse habitat characteristics produced variable effects and responses among the three Wyoming big sagebrush associations. On all associations habitat cover was reduced as result of complete to nearly complete loss of big sagebrush cover. The loss of big sagebrush

had a negative effect by not only decreasing structural cover but also reducing or eliminating forage provided by big sagebrush for sage-grouse. On the Thurber's-Idaho fescue association burning reduced Wyoming big sagebrush production by 450 kg ha⁻¹ (Davies and others 2007). The loss of big sagebrush cover on burned sites was not compensated by increases in tall herbaceous cover (>18 cm). As a consequence, vegetation cover requirements in the burns did not meet habitat guidelines for sage-grouse as suggested by Connelly and others (2000b).

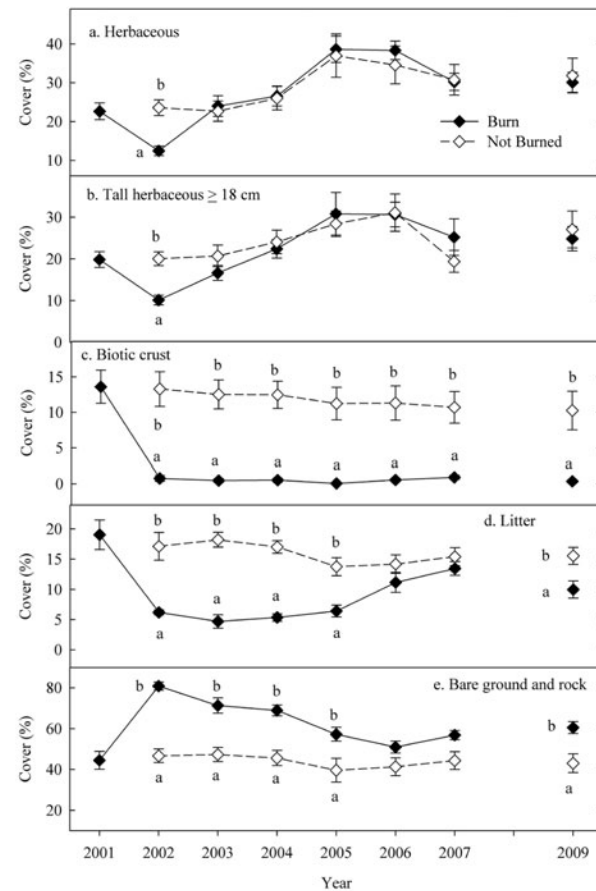


Figure 4. Ground covers (%) for the burned and not burned sites, Wyoming big sagebrush/bluebunch wheatgrass association, Sheephead Mountains, Oregon (2001-2009); (a) herbaceous cover, (b) tall herbaceous cover (>18 cm ht), (c) biotic crust, (d) litter cover, and (e) bare ground and rock. Data are in means $\pm$ SE. Means sharing a common lower case letter are not significantly different ($P>0.05$).

On two associations (Thurber's-Idaho fescue, Bluebunch) herbaceous composition and cover recovered by the second to third year after fire, and at present they have the appearance of grasslands. Although perennial grass cover did not differ between burned and not burned sites perennial grass yields

about doubled on burned sites. This discrepancy in perennial grass dynamics (yield and canopy cover) likely results from higher tiller density, greater reproductive effort, and taller plants on burned sites and also because canopy cover estimates are less precise than biomass measurements (Bates and others 2009). The 2 to 3 year recovery periods on these two associations are similar to herbaceous response after fire elsewhere in big sagebrush steppe (Blaisdell 1953; Conrad and Poulton 1966; Harniss and Murray 1973; Uresk and others 1976; West and Hassan 1985).

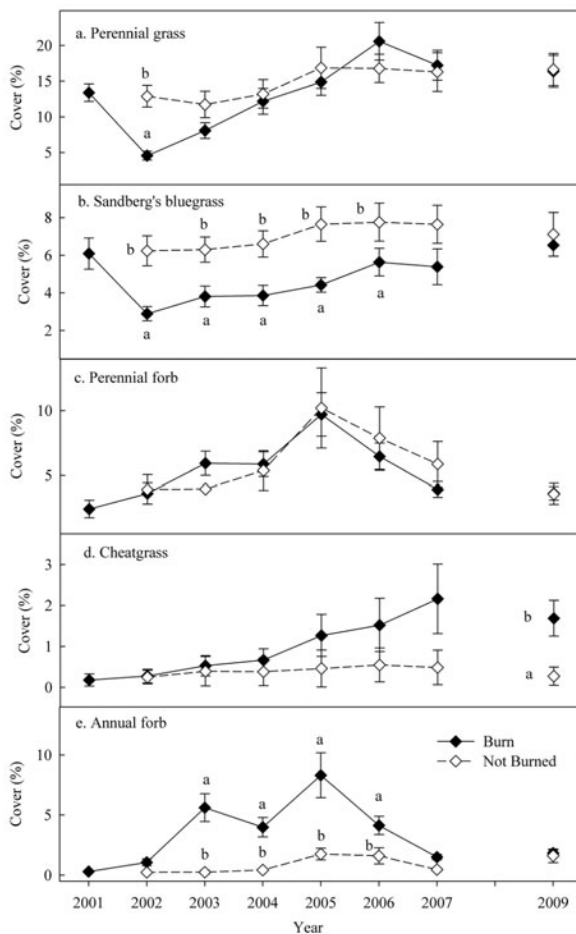


Figure 5. Herbaceous covers (%) for the burned and not burned sites, Wyoming big sagebrush/ bluebunch wheatgrass association, Sheephead Mountains, Oregon (2001-2009); (a) perennial grass (e.g. Thurber's needlegrass and other large bunchgrasses), (b) Sandberg's bluegrass, (c) perennial forb, (d) cheatgrass, and (e) annual forb. Data are in means $\pm$ SE. Means sharing a common lower case letter are not significantly different ($P > 0.05$).

Herbaceous recovery in these associations was mainly a result of low to no mortality among bunchgrass species and increases in cover of pale alyssum (Thurber's needlegrass-Idaho fescue association) or little blue-eyed Mary (Bluebunch association). The lack of major compositional changes in the bluebunch association and the Thurber's-Idaho fescue association indicated that they were resistant to summer wildfire and early fall prescribed fire, respectively. Their rapid recovery of herbaceous composition, cover, and productivity also indicates these associations were resilient following their respective fire disturbances.

The effects of fire on species in the Sheephead wildfire (table 3) generally agreed with impacts reported in the literature (Britton and others 1990; Conrad and Poulton 1966; Tisdale and 1969; Uresk and others 1976; Wright and Bailey 1982; Wright and others 1979). Mat-forming forbs and bunchgrasses with densely packed culms (Thurber's needlegrass, Idaho fescue, and Cusick's bluegrass) were the most severely impacted species. This was evident on the Thurber's needlegrass association where herbaceous composition has not recovered 8 years after wildfire. In the Bluebunch association, Idaho fescue and Cusick's bluegrass were almost eliminated; however, because these species were a minor component of pre-burn herbaceous cover (< 2 percent of total cover; < 1 plant/m²), there was little impact to overall herbaceous cover or composition.

In the Thurber's association the greater than 95 percent mortality suffered by perennial grasses resulted in little remaining tall herbaceous cover. High mortality of Thurber's needlegrass has been reported for summer wildfires (Uresk and others 1980; Wright and Klemmedson 1965). Herbaceous composition was largely comprised of native and non-native forbs and cheatgrass. Increases in total herbaceous cover were a result of increases in long-leaf phlox, cheatgrass, and annual forbs, particularly pale alyssum. These low growing species provide little value as escape or nesting cover for sage-grouse (Connelly and others 2000b; Crawford and others 2004). The compositional changes suggest that Thurber's needlegrass associations have potentially low resistance and resilience to wildfire. Prescribed fire on similar associations can have comparable effects when high litter accumulation in bunchgrass crowns increases burn residence times and causes

high bunchgrass mortality, resulting in site dominance by cheatgrass and other weedy species (Davies and others 2009).

Sagebrush Recovery

Recovery of Wyoming big sagebrush was limited on all associations after fires. Wyoming big sagebrush is the slowest of the big sagebrush species to recover after fire because of a lack of seed production in most years and because drier conditions make establishment of new plants problematic (Bates et al. 2005; Wright and Bailey 1982). Recovery periods for Wyoming big sagebrush after wildfire are not well quantified and have been variously estimated to take between 50 and 200 years (Baker 2006; Tisdale and Hironaka 1981; Wright and others 1979). Lesica and others (2007) measured a maximum of only 5 percent recovery of Wyoming big sagebrush canopy after a series of wildfires (time since fire, 7–23 yr) in southwestern Montana.

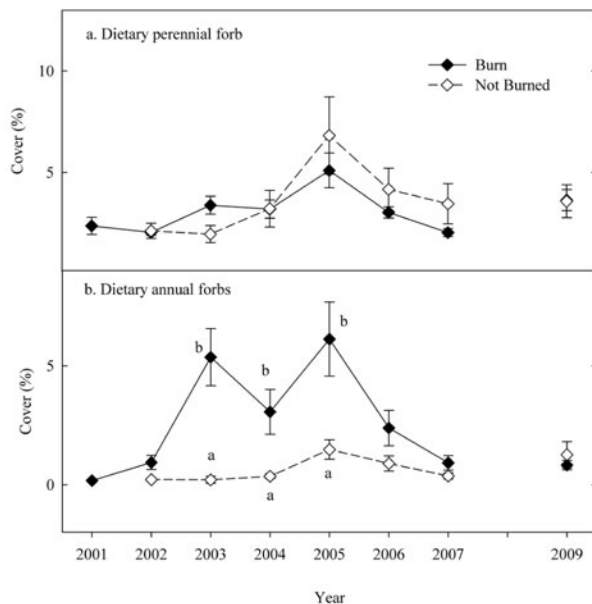


Figure 6. Cover of (a) dietary perennial and (b) dietary annual forbs that are utilized by sage-grouse, burned and not burned sites, Wyoming big sagebrush/ bluebunch wheatgrass association, Sheephead Mountains, Oregon (2001-2009). Data are in means $\pm$ SE. Means sharing a common lower case letter are not significantly different ($P > 0.05$).

Recovery of Wyoming big sagebrush after prescribed fire may occur earlier than after wildfire because fires are often less complete and retain pockets of surviving sagebrush. On the Thurber's-Idaho fescue association surviving sagebrush were scattered

throughout the burn and provide a potential seed source. Wyoming big sagebrush cover was about 10 percent of pre-burn levels on the Thurber's-Idaho fescue association seven years after fire. Longer term development on this site will likely result in greater landscape heterogeneity in the form of a grass and shrub mosaic which should benefit a greater variety of wildlife species. On other prescribed burns, Wambolt and Payne (1986) measured a 12 percent recovery of Wyoming big sagebrush cover 18 years after burning in southwest Montana; Beck and others (2009) reported that big sagebrush cover was 20 percent of preburn levels 14 years after prescribed fire; and Wambolt and others (2001) measured a 72 percent recovery 32 years after early fall prescribed fire.

Forb Response

A justification for burning and applying mechanical treatments in big sagebrush habitat has been to enhance forb abundance and productivity from sage-grouse pre-nesting through brood-rearing periods. The evidence indicates that there is limited potential for enhancing perennial forb yield or abundance after fire in Wyoming big sagebrush steppe. On the Thurber's needlegrass-Idaho fescue and bluebunch associations there was no increase in yield or cover of perennial forb species or genera reported to be important in the diet of sage-grouse. The only perennial forb to respond positively to fire was long-leaf phlox on the Thurber's association. Other studies have failed to detect any increase in forb diversity or abundance after burning or mowing in Wyoming big sagebrush communities (Beck and others 2009; Davies and others, in review; Fischer and others 1996; Wroblewski and Kauffman 2003). Crude protein of perennial forbs may be enhanced after fire; however, the effects are of short duration, lasting only the first two years after fire (Rhodes and others 2010).

In mountain big sagebrush communities burning will not necessarily result in substantial increases in perennial forbs. In eastern Oregon, frequency of Cichorieae species increased but abundance of other forbs consumed by sage-grouse did not after prescribed fire (Pyle and Crawford 1996). In southeastern Idaho, forb abundance across different-aged burns did not differ from unburned mountain big sagebrush communities (Nelle and others 2000).

On all associations cover/yields of annual forbs utilized by sage-grouse increased the first 1 to 3 years post-fire. On the Thurber's needlegrass-Idaho fescue and Thurber's association's annual forbs were mainly

comprised of pale alyssum by the second year after fire. Diet studies do not indicate that pale alyssum is consumed by sage-grouse (Barnett and Crawford 1994; Drut and others 1994; Klebenow and Gray 1968; Peterson 1970; Wallestad and others 1975). Sage-grouse utilize other annual mustards with a similar phenology (Peterson 1970); therefore, there is the potential that grouse may utilize pale alyssum.

Several factors account for the limited native forb response to fire including postfire weather, site potential, interference by perennial grasses and pale alyssum, and lack of forb propagules in the soil seed bank. The amount and timing of precipitation and temperatures can have a major influence on herbaceous productivity in big sagebrush steppe (Bates and others 2005; Sneva 1982). Perennial forbs on the Bluebunch association increased in cover in years with higher precipitation; however, the increase was identical on burned and not-burned sites. Weather did not influence perennial forb production in the Thurber's or Thurber's-Idaho Fescue associations because cover and/or yields did not differ across years despite 4 years of below-average precipitation and 2 years of above-average precipitation.

The potential for increasing perennial forbs may be limited by site characteristics. Perennial forb cover in most Wyoming big sagebrush associations comprises 15-22 percent of total herbaceous cover (Davies and others 2006). Production of perennial forbs and annual forbs in most Wyoming big sagebrush associations of Oregon average about 20 percent (15-30 percent) and 3 percent (0-8.5 percent) of total production, respectively (N=32, EOARC file data). Prior to fire, perennial forb cover and biomass represented 14 percent and 13 percent of total herbaceous cover and biomass, respectively, on the Thurber's needlegrass-Idaho fescue association. After fire the ratio of perennial forbs to total herbaceous production declined below 10 percent because perennial grass and pale alyssum yield increased and forb yield did not change. The response of perennial grasses and other herbaceous plant life forms after the fire may interfere with the ability of native forbs to increase after fire. Increased mortality of perennial grasses could increase the availability of openings for native forbs to establish. However, on the Thurber's needlegrass-Idaho fescue association increased mortality of perennial grasses would probably only have benefited pale alyssum rather than native forbs.

In addition, increased mortality of perennial bunchgrasses may also result in cheatgrass invasion or dominance, because this species is present within most Wyoming big sagebrush communities (Davies and others 2006; Davies and others 2008).

Fire in Sage-Grouse Habitat

Evaluating fire or other disturbances in ecosystems is challenging because the impacts can be construed positively, negatively, or without effect depending upon the objectives, scale, and duration of the analysis. In this study we evaluated the short-term impacts of fire on habitat characteristics of Wyoming big sagebrush steppe for sage-grouse, a sagebrush obligate. There is little indication that prescribed burning in Wyoming big sagebrush steppe will provide short-term benefits to sage-grouse. Burning big sagebrush steppe reduces potential nesting areas and roosting cover, as well as diminishing or eliminating forage provided by sagebrush for sage-grouse, which would be especially damaging in year-round and wintering habitat. Population studies indicate that sage-grouse numbers have declined following prescribed burning in Wyoming big sagebrush communities of southeast Idaho, which was likely a result of losses in nesting and wintering habitat (Connelly and Braun 1997; Connelly and others 2000a). In brood-rearing habitat an objective of prescribed fire has been to enhance forb production and availability (Wirth and Pyke 2003). In our studies and others (Beck and others 2009; Fischer and others 1996; Nelle and others 2000; Wroblewski and Kauffman 2003), yields or cover of forbs used by sage-grouse in their diets have been largely unresponsive to fire.

Insects are an important dietary component of young sage-grouse and may comprise 75-100 percent of the diet the first several weeks after hatching (Johnson and Boyce 1990; Patterson 1952). Juvenile sage-grouse survival was positively correlated with high Lepidoptera availability and frequency of slender phlox, and without insects in the diet, mortality rates of 90-100 percent in juvenile sage-grouse have been reported (Gregg 2006; Johnson and Boyce 1990). Fire was detrimental to ant populations in our study (Thurber's needlegrass-Idaho fescue association) and elsewhere in the Wyoming big sagebrush complex (Fisher and others 1996).

Aside from sage-grouse, there are several benefits after burning or mechanically treating big sagebrush for other wildlife species. For large ungulates and granivores, burned areas often result in a doubling of available herbaceous forage and may triple grass seed yield (Bates and others 2009; Cook and others 1994; Davies and others 2007). In mountain big sagebrush communities, creating a mosaic pattern of small treated areas within stands increased sage-grouse brood-rearing and summer use in Utah (Dahlgren and others 2006). A mosaic of different aged burns or greater habitat complexity in sagebrush steppe results in increased invertebrate biomass and avian species diversity and numbers (Petersen and Best 1987; Pons and others 2003; Noson and others 2006; Reinkensmeyer and others 2007; Roth 1976). The advantage of prescribed burning and mechanical brush control is that these treatments can be manipulated to construct a mosaic of sage-brush-steppe and herbaceous dominated areas.

In areas containing Wyoming big sagebrush communities, management of both wild and prescribed fires must be carefully considered. The high mortality of perennial grasses and presence of cheatgrass in the Thurber's needlegrass association suggests there is a substantial risk for annual grass replacement of this association after wildfire and potentially with prescribed burning (Davies and others 2008). Although the Sheepshead wildfire did not severely impact the herbaceous layer in the bluebunch wheatgrass association, this association and Thurber's needlegrass associations are often arranged in a mosaic across the landscape. These associations are the most commonly encountered type in Wyoming big sagebrush steppe of the northern Great Basin (Davies and others 2006). Thus, efforts should be made to limit wildfire disturbance in remaining Wyoming big sagebrush plant associations of the northern Great Basin. The danger of cheatgrass dominance is that wildfire frequencies are likely to increase compared to historic MFRI resulting in further degradation or loss of sagebrush communities (Whisenant 1990). In the Snake River Plains of Idaho, fires typically occur about every 5 years as a result of cheatgrass dominance in former Wyoming big sagebrush communities (Whisenant 1990). These fires are landscape level burns that limit recovery of big sagebrush and associated species (Suring and others 2005). Historically, the Wyoming big sagebrush cover type burned every 50–100 years and fires typically produced a mosaic of burned and

unburned patches (Wright and others 1979; West 1983; West and Hassan 1985).

MANAGEMENT IMPLICATIONS

The viability of obligate wildlife species, such as sage-grouse, is best served by maintenance of preferred habitat; thus disturbances, particularly those of large scale, are likely to have deleterious impacts to their populations. Because of the lack of quantifiable short-term benefits of burning in Wyoming big sagebrush communities it appears unnecessary to apply extensive or small-scale brush control treatments for specifically improving habitat for sage-grouse. Burning of Wyoming big sagebrush communities to enhance other species habitat requirements or to increase forage production for livestock should be applied when mortality of native perennial grasses and forb species can be minimized, result in a mosaic pattern of burned and unburned patches, and avoids areas where cheatgrass and other exotics are of concern. Furthermore, when planning prescribed fire in sagebrush steppe, areas of critical habitat should be identified, such as wintering grounds, in order to minimize potential negative impacts to sage-grouse and other sagebrush obligate and facultative wildlife species.

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Estimating Historical Sage-Grouse Habitat Abundance Using a State-and-Transition Model

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ABSTRACT

*Use of reference conditions to compare current conditions what managers believed represented healthy and functioning systems has become a common approach to evaluate vegetation and habitat conditions and aid development of land management plans. Often reference conditions attempt to describe landscapes as they existed and functioned prior to about 1850, and often largely rely on expert opinion. We developed reference conditions for sagebrush (*Artemisia* spp. L.) ecosystems in eastern Oregon based on ecological site descriptions, soil surveys, climate data, wildfire records, expert opinion, and literature using a state-and-transition (STM) modeling framework. Using ecological site descriptions for the Malheur High Plateau Major Land Resource Area (MHP), we divided sagebrush communities into four groups based on grass productivity in low, average and high productivity years. Literature helped us determine which disturbance factors to include, the community phases for each model, and associated seasonal habitat for greater sage-grouse (*Centrocercus urophasianus*). We developed successional timelines in the absence of disturbance, and determined the probable outcomes of a given type of disturbance event. We used fire records and climate data to develop disturbance event probabilities and periodicities. Contrary to our expectations, fire did not appear to be the most important factor influencing sagebrush ecosystems under reference conditions in our models. The modeled historical abundance of sage-grouse breeding and brood-rearing habitat was within range of or greater than the amount recommended by sage-grouse biologists, but the abundance of wintering habitat was less. By using objective criteria as much as possible, our approach should also be repeatable in other locations. Since we used climate criteria to define most disturbance probabilities, our models provide an opportunity to examine how changes in climate could affect plant communities, disturbance regimes, and the quality and quantity of sage-grouse habitat in future modeling efforts.*

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INTRODUCTION

Sagebrush (*Artemisia* spp. L.) ecosystems provide many important economic and social values in the Intermountain West, such as livestock forage, water, recreational opportunities, and habitat for a variety of wildlife species. Changes to sagebrush ecosystems over the last 150 years threaten their ability to provide many of these values in the future (Connelly et al. 2004; Miller and Eddleman 2000). Human-related disturbances, invasive species, expansion of conifer woodlands, changes in fire regimes, and changes in climate have all been involved in reducing the area occupied by sagebrush ecosystems by an estimated 14.8 million ha across the western United States

(USDI BLM 2004). Habitat for the greater sage-grouse (*Centrocercus urophasianus*), a candidate species for listing under the Endangered Species Act, is of great concern in many areas of the interior West (Bunting et al. 2002; Connelly et al. 2004; Knick et al. 2003).

Under current ecosystem management practices, federal land managers compare current conditions to reference conditions to evaluate changes in land health and probable causes of those changes. Generally, reference, or historical, conditions are based on some measure or description of conditions present around 1850 in the western United States. However, the lack of detailed descriptions and

suitable surrogates, such as tree ring studies, and the lack of stand or patch-scale vegetation modeling tools in rangelands mean that expert opinion often forms a large part of the basis for the reference condition descriptions. In the absence of intact reference areas to serve as a basis, different experts may form very different opinions of the reference conditions and what factors were important in creating those conditions.

State-and transition modeling frameworks (STMs), such as the Vegetation Dynamics Development Tool (VDDT) (ESSA Technologies Ltd. 2007), offer the promise of developing reference conditions that are more objective and repeatable, using a process that is transferable to other landscapes. These modeling frameworks can be used at a scale suitable for land use planning, can incorporate management actions and relevant natural disturbances, and fit directly with current rangeland ecology paradigms (Briske et al. 2006; Stringham et al. 2003; Westoby et al. 1989). Since STMs are probabilistic instead of mechanistic, they can operate with a combination of empirical data and expert opinion where empirical data are lacking; a common condition in rangeland management. Climate variables can form the basis of event probabilities to predict plant community changes.

Our goals in this study were to evaluate the use of climate variables as a basis for event probabilities and to evaluate how historical disturbances may have influenced reference conditions in sagebrush communities, with an emphasis on the quantity and quality of sage-grouse habitat. Our primary objective was to develop VDDT-based models that could 1) simulate the effects of natural disturbances on plant community dynamics using fire, soils, and climate data, 2) incorporate available information from the scientific literature on sagebrush-steppe ecosystems, and 3) use selected rules used in mechanistic vegetation models. We used the literature, climatic records, and a limited amount of expert opinion to develop probabilities of disturbance and successional pathways and rates for four sagebrush groupings. Sagebrush groupings were based on ecological site descriptions. We estimated the amount of each community phase and the resulting quantity of sage-grouse habitat within each sagebrush group and for the landscape as a whole. Terminology follows that used by the state-and-transition literature (Bestelmeyer et al. 2009).

STUDY AREA

The study area was the 4-million ha Malheur High Plateau (MHP) major land resource area (NRCS 2006) in southeastern Oregon (figure 1). Much of the area lies between 1190 m and 2105 m elevation, with Steens Mountain reaching 2967 m. The area contains no major rivers and little surface water but has numerous springs, shallow lakes, and playas. Perennial streams and small rivers are mostly located on the periphery. Using soil series descriptions (<http://soils.usda.gov/technical/classification/osd/index.html>) we estimated that 98 percent of the soils in the sagebrush ecological types of the MHP are Mollisols and Aridisols. Soils are primarily loamy to clayey, well-drained and shallow (25 to 50 cm) to moderately deep (50 to 90 cm) in uplands, and poorly to well-drained and deep to very deep (>90 cm) in basins.

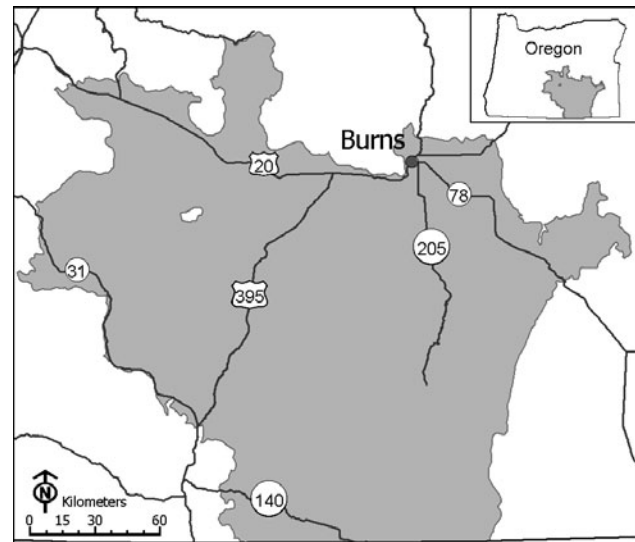


Figure 1. Location of the study area in Oregon. The High Desert Province Ecological Province and the Malheur High Plateau Major Land Resource Area occupy approximately the same area, with the exception of the area to the east of Steens Mountain. The Malheur High Plateau Major Land Resource Area includes some area to the east of Steens Mountain while the High Desert Ecological Province does not. The area to the east of Steens Mountain lies within the rain shadow of the mountain and has a different climate. The town of Burns is the largest community within the study area.

The average annual precipitation ranges from 105 mm to 305 mm over most of the area. Winter and spring are the wettest periods with most precipitation falling in November, December, January and May, while summer is the driest. January is the coolest

month, averaging -2°C , and July the warmest, averaging 19°C . Sagebrush-steppe (*Artemisia* spp. L. and caespitose grasses) is the dominant vegetation type with salt desert shrub (*Sarcobatus vermiculatus* (Hook.) Torr.-*Grayia spinosa* (Hook.) Moq.) on saline soils in basins, western juniper (*Juniperus occidentalis* Hook. var. *occidentalis*) expanding out from rockier areas, and aspen (*Populus tremuloides* Michx.) at the higher elevations.

Model Design and Assumptions

We selected 1350 to 1850 as our historical reference period, a period commonly known as the Little Ice Age. Although the climate was cooler and wetter than present, it had shifted into a winter-dominant precipitation regime with plant communities very similar to present (Miller and Wigand 1994). Prior to this period, climate was warmer and drier than present with less dominance of winter precipitation and different disturbance regimes (Cook et al. 2004; Miller and Wigand 1994).

We used instrument-based climate records to develop rules for determining the frequencies of climate-related events (Neilson et al. 1992), using these frequencies in combination with other information sources and expert opinion to estimate the probabilities of several disturbance types and establishment rates for sagebrush. Data sources included temperature and precipitation records for Oregon Climate Division 7 (OCD7) (<http://www7.ncdc.noaa.gov/CDO/CDODivisionSelect.jsp>) organized by water year (October through September) for 1894 to 2007; snow data from the Reynolds Creek Experimental Range for 1967 to 1996 (Hanson et al. 2001; Marks et al. 2001), and local remote area weather stations (RAWS) (<http://www.raws.dri.edu/index.html>). Although the Reynolds Creek Experimental Range lies outside the study area, the climate is similar (Hanson et al. 2001) and detailed snowfall data are available for this location that are not available for OCD7.

We used ecological site descriptions for the MHP (<http://esis.sc.egov.usda.gov>) to divide the sagebrush-grass plant communities into four groups based on grass productivity in low, average, and high production years. We designated these groups as Warm-Moist Sagebrush (WM Group), Cool-Moist Sagebrush (CM Group), Warm-Dry Sagebrush (WD Group), and Shallow-Dry Sagebrush (SD Group). Since site productivity influences recovery rates

following fire (Bollinger and Perryman 2008; Boltz 1994; Lesica et al. 2007; Wambolt et al. 2001), we assumed the same applied equally well to other disturbances. We used grass production of 672 kg ha^{-1} as the threshold for these divisions since that level of production is considered the minimum needed to support fire spread in bunchgrass fuels under moderate burning conditions (Bunting et al. 1987; Gruell et al. 1986).

The WM Group, the most productive group, typically resided on xeric, mesic, deep to very deep soils. Water storage capacity was high and many sites were subirrigated. This group occurred mostly in swales, terraces, and near or in riparian areas below 1220 m elevation. It occupied an estimated 11 percent of the MHP, based on soil surveys (http://www.or.nrcs.usda.gov/pnw_soil/or_data.html). The modal community was basin big sagebrush (*Artemisia tridentata* Nutt. ssp. *tridentata*)/basin wildrye (*Leymus cinereus* (Scribn. & Merr.) A. Löve). We included fire, drought, and insects as the important disturbances in this group.

The CM Group was found on xeric, frigid, moderately deep to deep soils mostly above 1220 m elevation. Soils had a high water storage capacity, but subirrigation was rare to nonexistent. This group typically occurred on ridges, northerly aspects at lower elevations, and all aspects at higher elevations, and occupied an estimated 16 percent of the MHP. The modal sagebrush community was mountain big sagebrush (*Artemisia tridentata* Nutt. ssp. *vaseyana* (Rydb.) Beetle)/Idaho fescue (*Festuca idahoensis* Elmer). We included fire, drought, insects, freeze-kill, snow mold, and voles as major disturbances.

The WD Group was found on aridic, mesic, moderately deep to shallow soils up to 1400 m elevation. Water holding capacity was moderate to low and sites tended to become quite dry by mid to late summer. This group occurred mostly on southerly aspects at higher elevations, well-drained soils, and relatively shallow soils in basin bottoms and terraces, and occupied approximately 61 percent of the province. The modal plant community was Wyoming big sagebrush (*Artemisia tridentata* Nutt. ssp. *wyomingensis* Beetle & Young)/bluebunch wheatgrass (*Pseudoroegneria spicata* (Pursh) A. Löve)-Thurber's needlegrass (*Achnatherum thurberianum* (Piper) Barkworth). Factors included in this group were fire, drought, insects, and pronghorn browsing.

The SD Group, the least productive sagebrush environment, occupied aridic, mesic to frigid, shallow to very shallow soils at any elevation. Soils typically had low water storage capacity and high evaporation rates from temperature, wind, or both and became quite dry by late spring or early summer. The SD Group covered an estimated 12 percent of the MHP. The modal plant community was low sagebrush (*Artemisia arbuscula* Nutt.)/Sandbergs bluegrass (*Poa secunda* J. Presl). Factors included in this group were fire, drought, insects, and pronghorn browsing.

We built STMs for all four groups using VDDT version 6.0.9 (ESSA Technologies Ltd. 2007). All models used four community phases (figure 2). Grasses and forbs dominated the early seral (ES) community phase. In the midseral open (MSO) phase, mature sagebrush was present but ecologically subdominant, and grasses and forbs were dominant. Sagebrush, grass and forbs co-dominated in the late seral open (LSO) community phase. Sagebrush was dominant in the late seral closed (LSC) community phase. We used sagebrush cover as the indicator of movement from one community phase to the next.

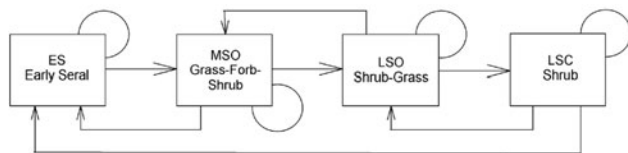


Figure 2. Model structure. Arrows pointing to the right indicate deterministic transitions resulting from succession. Arrows pointing to the left indicate probabilistic transitions to an earlier community phase. Circles indicate probabilistic transitions that remain in the same community phase. ES = early seral, MSO = midseral open, LSO = late seral open, LSC = late seral closed.

We assumed sagebrush density and cover were initially low following a high severity disturbance then increased until the site reached full occupancy (Daubenmire 1975; Harniss and Murray 1973; Johnson 1969; Lesica et al. 2007; Perryman et al. 2001) and that soil moisture availability in spring and early summer were key to sagebrush establishment (Boltz 1994; Daubenmire 1975; Johnson and Payne 1968; Lomasson 1948; Meyer 1994). Sagebrush establishment in the CM, WD, and SD groups was based on various combinations of spring precipitation, temperature and season length intended to represent adequate soil moisture. We assumed establishment in the WM Group was based on random weather factors

we could not assess through the available data, such as the specific timing of precipitation events and any heat waves or cold snaps. We estimated the time needed to reach sagebrush cover breakpoints between each community phase based on sagebrush crown measurements and growth rates from published studies involving common gardens and wild plants (Anderson and Inouye 2001; Johnson 1969; McArthur and Welch 1982; Miller and Eddleman 2000; Pringle 1960; Tisdale et al. 1965; Wambolt and Sherwood 1999; Wambolt et al. 2001; Winward 1991).

An extensive review of the sagebrush and wildlife literature combined with preliminary model testing indicated we should include fire (Connelly et al. 2004; Knick et al. 2003; Knick et al. 2005), drought (Allred 1941; Ellison and Woolfolk 1937; Pechanec et al. 1937), freeze-kill (Hanson et al. 1982; Walser et al. 1990), snow mold (Nelson and Sturges 1986; Sturges and Nelson 1986; Sturges 1986, 1989) and herbivory as major disturbances. Native herbivores of most importance to local sagebrush ecosystems included pronghorn (*Antilocapra americana*) (Hansen and Clark 1977; Howard 1995; MacCracken and Hansen 1981; Verts and Carraway 1998), voles (*Microtus* spp.) (Hubbard and McKeever 1961; Mueggler 1967), and several species of insects (Allred 1941; Gates 1964; Hall 1965; Welch 2005) of which aroga moth (*Aroga websteri* Clark) appeared to be the most ecologically significant.

We used monthly or seasonal temperature, precipitation or snow depth to estimate probabilities for fire, freeze-kill (DeGaetano and Wilks 2002; Hanson et al. 1982; Hardy et al. 2001, Walser et al. 1990), snow mold (Nelson and Sturges 1986; Sturges and Nelson 1986; Sturges 1989), severe pronghorn browsing (Bilbrough and Richards 1993; Hoffman and Wambolt 1996; McArthur et al. 1988, Smith 1949), and vole-related sagebrush mortality (Frschknecht and Baker 1972; Mueggler 1967; Parmenter et al. 1987). We created variability modifiers for fire and pronghorn impacts by estimating the percentage of years in different severity categories (low, average, high, and extreme), the average number of hectares per event in each severity category, and the ratio of hectares affected in each severity category. We based fire variability on the variability of fire season severity in modern fire records from Bureau of Land Management (BLM) for Burns and Lakeview Districts and from the US Fish and Wildlife Service (FWS) for

Hart Mountain Refuge. Pronghorn variability was based on a very simple model of pronghorn population dynamics to estimate the frequency of population peaks and lows based on winter conditions (Kindschy et al. 1982; O'Gara and Yoakum 2004; Smyser et al. 2006; Yoakum 2006).

We reduced climate-based estimates of fire occurrence to account for the lack of ignitions when sufficient fuel is present. We also partitioned fire into two different burn patterns – a mosaic (heterogeneous) burn pattern and a stand-replacing (homogeneous) burn pattern. These burn patterns are approximate equivalents of mixed severity and high severity fires in forests. We assumed homogeneous fires resulted from high winds and used frequency of high winds in August based on hourly data from local RAWS to estimate the occurrence of homogeneous burn patterns. We then assumed that heterogeneous burn patterns occur in low, average, and high years, and homogeneous burn patterns occur in high and extreme years. Once a site reached the LSC phase, only homogeneous fire occurred to account for the effects of sagebrush density and cover on fine fuel

abundance (Bradford and Laurenroth 2006; Daubenmire 1975; Derner et al. 2008).

We based drought probability on the estimated frequency of droughts as severe as that in the 1930s (Cook et al. 1999; Cook et al. 2004; Gedalof and Smith 2001; Graumlich 1987; Stahle et al. 2007), the only drought with documented big sagebrush mortality (Allred 1941; Ellison and Woolfolk 1937; Pechanec et al. 1937). Insect outbreak frequencies were based on a forest defoliator as a surrogate due to the lack of detail on aroga moth dynamics, the primary insect affecting sagebrush (Gates 1964; Hall 1965; Hsaio 1986). We selected Pandora moth (*Coloradia pandora* Blake) to represent probable frequencies and variability (Gates 1964; Hall 1965; Hsaio 1986; McBrien et al. 1983; Speer et al. 2001). We used a combination of the vole population cycle (Frischknecht and Baker 1972; Murray 1965) and frequency of severe winters (Frischknecht and Baker 1972; Parmenter et al. 1987) to estimate the probability of vole-related mortality. Because vole populations also depend on the abundance of grass, we varied the probability of vole impacts by community phase.

Table 1. Habitat suitability (low, moderate, high) for greater sage-grouse by model and community phase based on descriptions from Call and Maser 1985; Connelly et al. 2000, 2004; Crawford and Gregg 2001; Goodrich 2005; Braun et al. 2005; and Gregg 2006.

	Leks	Pre-laying hens	Nesting	Early brood- rearing	Late brood- rearing	Wintering
Warm moist sagebrush group						
ES ^a	Low	Moderate	N/A	Low	Moderate	N/A
MSO ^b	N/A	Moderate	Low	Moderate	High	N/A
LSO ^c	N/A	Low	High	High	High	High
LSC ^d	N/A	N/A	Moderate	Low	Low	Moderate
Cool moist sagebrush group						
ES	Low	High	N/A	Low	Moderate	N/A
MSO	N/A	High	Low	Low	High	N/A
LSO	N/A	Moderate	High	High	High	High
LSC	N/A	N/A	Moderate	Moderate	Moderate	Moderate
Warm dry sagebrush group						
ES	Moderate	High	N/A	Low	Low	N/A
MSO	Low	High	N/A	High	Low	Low
LSO	N/A	Moderate	High	High	Low	High
LSC	N/A	Moderate	Moderate	Moderate	Low	High
Shallow dry sagebrush group						
ES	High	High	N/A	High ^e	Moderate	High ^f
MSO	Moderate	High	N/A	High ^e	Moderate	High ^f
LSO	Low	High	N/A	High ^e	Low	High ^f
LSC	N/A	High	N/A	High ^e	Low	High ^f

^a Early seral. ^b Midseral open. ^c Late seral open. ^d Late seral closed. ^e High along edges, dropping to low in interior. ^f High until or unless buried by snow.

Table 2. Ranking of disturbance types in each sagebrush model based on the estimated disturbance rotation period.

Warm moist sagebrush group	Cool moist sagebrush group	Warm dry sagebrush group	Shallow dry sagebrush group
Insects	Snow mold	Pronghorn	Pronghorn
Fire	Voles	Insects	Insects
Drought	Insects	Fire	Fire
	Freezekill	Drought	Drought
	Fire		
	Drought		

We used the descriptions of the different types of sage-grouse habitat provided by Connelly et al. (2000) to evaluate the potential effects of the disturbance variables on sage-grouse habitat suitability. Breeding habitat included lekking, pre-laying hen, and nesting habitat, and brood-rearing habitat included early and late habitats. We based habitat quality ratings on similarity to described habitat characteristics (Barnett and Crawford 1994; Braun et al. 2005; Call and Maser 1985; Connelly et al. 2004; Connelly et al. 2000; Crawford and Gregg 2001; Goodrich 2005; Gregg 2006). Each community phase was rated as none, low, moderate, or high quality habitat for each seasonal habitat based on sagebrush cover, assumed forb abundance and timing of senescence, and expected duration of the habitat in the absence of disturbance (table 1). We then summarized the amount of moderate- and high-quality seasonal habitat available for each group and habitat element and the four groups collectively.

Analysis Methods

Each model began with an equal proportion of the four community phases. We ran each model 50 times for 500 years and recorded the abundance of each community phase every 10 years. To allow ample time for the models to come into dynamic equilibrium, we analyzed only the last 250 years of data. We conducted sensitivity tests to evaluate how the mix of community phases might change if we altered event probabilities from those initially developed. After finalizing the models based on the sensitivity testing, we estimated the amount of historical seasonal habitat for sage-grouse in each sagebrush group and on the landscape as a whole and compared the results to the amount of habitat recommended by Connelly et al. (2000). We compared the predicted fire rotation in models to estimated fire frequencies published in the literature. Because community phases were not normally distributed in most cases we analyzed medians rather than means.

Fire, drought, and insect outbreaks affected the full area occupied by the sagebrush groups. Freezekill, snow mold, and vole-related mortality occurred where snowpacks are deeper and more persistent (Frischknecht and Baker 1972; Hanson et al. 1982; Mueggler 1967; Nelson and Sturges 1986; Parmenter et al. 1987; Sturges and Nelson 1986; Sturges 1989; Walser et al. 1990), limiting them to a portion of the CM Group. Wintering pronghorn tended to move to where snowpacks were lowest and preferred habitat with long sightlines (Kindschy et al. 1982; O'Gara 1978; Verts and Carraway 1998); therefore, we assumed pronghorn impacts were restricted to a portion of the WD and SD Groups. We modeled fire, pronghorn browsing, freezekill, and snow mold as random events and drought, insects, and voles as cyclical events.

Our models accounted for the impacts to sagebrush only and not to other species or life forms. Homogeneous fire was the only stand-replacing event in all models, resetting any community phase back to ES. All other events were modeled as thinning events, resetting a community phase back to its beginning or moving it back one or two community phases. All thinning events operated only in the MSO, LSO, and LSC community phases. Insect outbreaks occurred only in the LSO and LSC phases in all models. Fire was the only event in the ES phase in all models.

RESULTS

Contrary to our expectations and based on average annual percentage of area affected, fire appeared to have less influence than other disturbance types, except drought (table 2). Estimated fire rotations were 24, 33, 83, and 196 years for the WM, CM, WD, and SD Groups, respectively. Most disturbance types occurred more frequently than fire. In each model, some sort of disturbance occurred rather frequently across the landscape as a whole. Frequencies ranged from every four years in the CM Group to every 26 years in the SD Group.

The LSC community phase was the most common phase in all groups under simulated historical conditions (figure 3). The least common phase was the MSO phase in the WM, WD, and SD Groups and the ES phase in the CM Group. All groups were sensitive to changes in the probability of fire and insects. The WM and CM Groups were insensitive to changes in the probability of drought, while the WD and SD Groups were sensitive. The CM Group also was moderately sensitive to changes in the probabilities of insect and vole outbreaks, and sensitive to changes in the probabilities of snow mold and freeze-kill, affecting the abundance of the MSO and LSC community phases more than the LSO phase in all cases except snow mold. Both the WD and SD Groups were sensitive to changes in the probability of pronghorn browsing. In general, increasing the probability of a disturbance tended to decrease the abundance of the later community phases and increase the abundance of the earlier community phases while reducing the probability had the opposite effect.

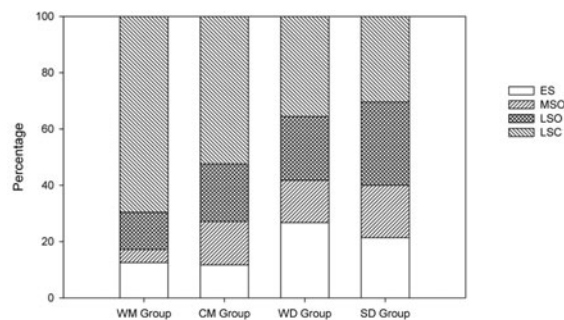


Figure 3. Mix of community phases. The late seral closed (LSC) phase is the most common in all models, although more dominant in the warm, moist (WM) and cool, moist (CM) models. The midseral open (MSO) phase is the least common in the WM, warm, dry (WD), and shallow, dry (SD) groups while the early seral (ES) phase is the least common in the CM group.

Altering the frequency of the different types of fire years had a large impact on fire rotation and the mix of community phases, particularly in the abundance of the ES phase, in all four groups. Natural fire rotation lengthened 2.7 times in the WM and CM Groups and 3.5 times in the WD and SD Groups. The resulting fire rotations were well outside the fire frequencies or rotation reported in the literature (Baker 2006; Burkhardt and Tisdale 1976; Heyerdahl et al. 2006; Knick et al. 2005; Mensing et al. 2006; Miller et al.

2001; Miller and Heyerdahl 2008; Miller and Rose 1999; Whisenant 1990).

The simulated historical landscape provided breeding habitat on 86 percent of the area, compared to the 80 percent recommended (Connelly et al. 2004), but only about one-quarter of that was high quality habitat. Brood-rearing habitat occurred on 64 percent of the landscape, with twice as much early brood-rearing habitat as late brood-rearing (figure 4). Most of the brood-rearing habitat was moderate quality. Wintering habitat was found on 53 percent of the simulated historical landscape with over half in the WD Group. We did not include early brood-rearing provided by the SD Group in these results as chicks use the edges of this habitat more than the interior (Aldridge 2000, 2005; Goodrich 2005) and we did not model patch shape or edge characteristics. Similarly, we did not include the SD Group in the wintering habitat total as that group provides habitat only in low snow years.

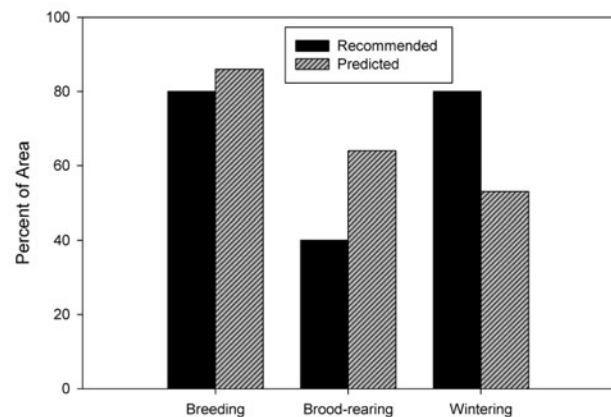


Figure 4. Amount of moderate and high quality seasonal sage-grouse habitat. Landscape amounts are based on the proportions of each group as determined from soil surveys in the Malheur High Plateau major land resource area. Not all groups provide all types of seasonal habitat.

DISCUSSION

Our results suggest that the various thinning agents, which are not as obvious as fire and not monitored for frequency, variability, or impacts, may have been more important than fire in affecting sage-grouse habitat historically. The current perception of the importance of fire on sage-grouse seasonal habitat may be based more on the current predominance of very large, homogeneous fires and current problems

with annual grasses that can follow such fires. Historically, insect outbreaks in particular may have been of equal importance as fire in shaping the abundance and quality of seasonal sage-grouse habitat. Insect outbreaks tend to affect a many-fold larger area when they occur (Gates 1964; Hall 1965) than the most severe fire season on record and may have occurred more frequently than fire in the WD and SD groups. These two groups comprise the majority of potential sage-grouse habitat in the study area.

We suspect that disturbance probabilities for some factors, such as fire, should vary by community phase, which could also influence the interactions between disturbances. For example, abundance and continuity of grasses and the relative proportion of live and dead woody fuel in sagebrush crowns likely varies between the different community phases in each model. This variation should affect the likelihood that fire could successfully ignite and spread. However, we lack sufficient information on that variation to adjust the probability of fire accordingly. Similarly, the amount of sagebrush cover would likely result in differing probabilities of insect outbreaks between community phases. We were able to estimate different probabilities by community phase only for voles, based on the winter diet of voles and relative proportion of sagebrush to grass in the CM Group (Mueggler 1967; Parmenter et al. 1987; Sturges 1993). In that model, it appeared the frequency of insect outbreaks altered the frequency of vole outbreaks by altering the abundance of the LSO community phase—the phase in which a vole outbreak is most likely to have an effect. If we were able to make similar distinctions in disturbance probabilities by community phase, then more interactions between disturbances might have occurred.

We speculate that modern burned-hectare totals per fire season in our study area may not be much different from those prior to 1850. Use of fire by Native Americans is well documented even in the Great Basin (Griffen 2002; Gruell 1985; Robbins 1999; Stewart 2002). Tree-ring studies of fire extent in pre-1850 forests indicate that regional fire years (years where fire is widespread throughout a large area, the equivalent of extreme fire years today) occurred at about the same frequency prior to 1850 as in modern fire records (Hessl et al. 2004;

Heyerdahl et al. 2008; Swetnam and Betancourt 1998). One possible difference between the 500 years before 1850 and the time since 1980 is average fire size, as compared to total hectares burned per year. Before 1850, a year where a great many hectares burned may have consisted of a large number of small to medium-sized fires. Since 1980, such years typically consist of a few very large fires, believed to be largely due to changes in fuel structure resulting from a variety of human-caused changes (Connelly et al. 2004; Heyerdahl et al. 2006; Knick et al. 2005). The landscape patterns and resulting sage-grouse habitat quality and availability would have been very different before 1850 than since 1980 even if the frequencies of the different types of fire years were similar.

We assumed if the fire frequencies in the literature and fire rotations from the models were relatively close, the model results were a reasonable representation of the reference period, predicting the mix of community phases and sage-grouse seasonal habitat. Therefore, we compared the estimated fire rotation in our final models against tree-ring based estimates and published expert opinion estimates of fire frequency. Tree-ring studies at the sagebrush-conifer ecotone indicate an average fire return interval of 10 to 35 years (Burkhardt and Tisdale 1976; Heyerdahl et al. 2006; Miller et al. 2001; Miller and Heyerdahl 2008; Miller and Rose 1999). Expert opinion for fire return intervals range from 10 to 25 years on more productive sites, 30 to 80 on less productive sites, and over 100 years on very dry, low-productivity sites (Knick et al. 2005; Miller and Heyerdahl 2008; Miller and Rose 1999). The modeled fire rotations all fall within these general categories. Thus, the indirect evidence suggests the mix of community phases is reasonable.

The simulated historical quantity of sage-grouse seasonal habitats appears to be similar to that recommended by sage-grouse biologists, with the exception of wintering habitat (Connelly et al. 2004; Connelly et al. 2000). Our models predicted that the MHP provided 6 percent more breeding habitat, about 50 percent more brood-rearing habitat, but around 34 percent less wintering habitat that sage-grouse biologists recommend (Connelly et al. 2000). Although sage-grouse will winter in the SD Group in many locations, the majority of wintering populations in Oregon have been observed in sites dominated by

big sagebrush (Connelly et al. 2004), placing them in either the WD or CM Groups. Assuming our model design was appropriate, the results suggest either the lower availability of wintering habitat might have been population bottleneck, or that sage-grouse did not need quite as much wintering habitat as biologists recommended.

CONCLUSIONS

This project demonstrates methods to examine potential sagebrush ecosystem dynamics and habitat for historical conditions using a state-and-transition modeling framework. It also demonstrates how climate data can be used to develop objective disturbance probabilities. Our study also provides objective criteria that could be used to evaluate expert opinion and the logical arguments that underpin such opinion. It also points out the importance of understanding the frequency and intensity of disturbance variables incorporated into such models.

The modeled fire rotations were within the range reported largely based on expert opinion in areas where surrogates for fire history are not available. Fire may not have been the most important disturbance factor shaping historical landscape patterns and habitat availability, just the most visible and easily studied factor. The frequency of the different types of fire season is an important, but possibly overlooked factor in how fire might have shaped historical habitat availability.

Sage-grouse breeding and brood-rearing habitat availability may have been greater than that recommended by sage-grouse biologists, but wintering habitat may have been less in the historical landscape. If so, these shortage categories along with a predominance of less than optimal habitat may indicate population bottlenecks that could have limited sage-grouse population potential. Disturbances that promote later community phases increase the abundance of nesting and wintering habitat. Disturbances that favor early phases increase lekking and pre-laying hen habitat, while disturbances that favor middle community phases increase brood-rearing habitat. Higher quality sage-grouse habitat across the landscape requires a mix of all community phases distributed among the four sagebrush groups.

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Strategic Use of Forage Kochia (*Kochia prostrata*) to Revegetate Wildlife Habitat

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ABSTRACT

Forage kochia (*Kochia prostrata* [L.] Schrad.) is a long-lived, perennial, half-shrub adapted to the temperate, semiarid regions of central Asia and the western U.S. In these areas it is a valuable fall/winter forage plant for sheep, goats, camels, cattle, horses, and wildlife. *Forage kochia* is extremely drought, heat, and salt tolerant. *Forage kochia* plants are very competitive with the annual noxious weeds cheatgrass (*Bromus tectorum* L.) and halogeton (*Halogeton glomeratus* [Stephen ex Bieb.] C.A. Mey.) and it is one of few species that can be successfully established on severely degraded, frequently burned, cheatgrass-infested rangelands. *Forage kochia* also is being used to establish 'greenstrips' to stop the spread of wildfires, due to its high moisture content and ability to reduce the frequency of highly flammable cheatgrass. *K. prostrata* and *K. scoparia* are both sometimes referred to as 'forage kochia' and 'summer cypress'; however, *K. prostrata* differs in that it has a perennial growth habit, does not spread into perennial plant stands, is not known to contain toxic levels of nitrates or oxalates, and increases biodiversity on rangelands. The cultivar 'Immigrant' was released in 1984 and remains the only released cultivar of *forage kochia* in the U.S., and is a short-statured, diploid type, used for livestock and wildlife forage, rangeland reclamation, and suppression of wildfires. An active breeding program is underway to develop larger statured, more productive *forage kochia* cultivars to enhance its utilization as winter forage and habitat in the temperate deserts of the western U.S. Overall, *forage kochia* is not likely to become a noxious weed, but does have the potential to improve the sustainability of rangelands and wildlife habitat in semiarid regions that frequently experience extended drought, salinity, and wildfires.

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INTRODUCTION

Forage kochia (*Kochia prostrata* [L.] Schrad.) (synonym=*Bassia prostrata* [L.] A.J. Scott), sometimes called prostrate kochia, or prostrate summer cypress is a long-lived, perennial, semi-evergreen, half-shrub adapted to semiarid, arid, alkaline, and saline rangelands and steppes (Harrison et al. 2000; Gintzburger et al. 2003). *Forage kochia*'s native distribution ranges from central Europe to the west, Siberia in the north, Afghanistan and Asia Minor in the south, and east to China, Mongolia, and Tibet (Balyan 1972). *Forage kochia* was introduced to North America in the early 1960s where it has proven to be well adapted to the temperate, semiarid rangelands of the western U.S. (Harrison et al. 2000). *Forage kochia* has a perennial woody base with yearly herbaceous growth that can reach heights ranging from 30 to 75 cm, and a thick woody root system that reaches depths of 3 to 6.5 m (Gintzburger et al. 2003).

Forage kochia is a distant relative of annual kochia (*K. scoparia* L.) and gray molly (*K. Americana* S. Wats), with recent research showing that these three species of *Kochia* are genomically distinct and do not cross hybridize (Lee et al. 2005). *K. prostrata* and *K. scoparia* are both sometimes referred to as 'forage kochia' and 'summer cypress'; however, *K. prostrata* differs in that it has a perennial growth habit, does not spread into perennial plant stands, and is not known to contain toxic levels of nitrates or oxalates (Harrison et al. 2000). *Forage kochia* (*K. prostrata*) is a complex species within the *Chenopodiaceae* family represented by multiple ploidy levels and subspecies (Balyan 1972; Gintzburger et al. 2003; Waldron et al. 2005).

Because of its competitive nature, some people have worried about *forage kochia* invading and suppressing native plant populations in the U.S. (Clements et al. 1997; Harrison et al. 2000). However, several researchers have reported that Immigrant *forage kochia* competes well with annuals, but does not invade perennial plant communities (Pendleton et al. 1992; Harrison et al. 2000; Monaco et al. 2003).

Competition With Annual Grasses and Stopping Wildfire

It has been shown that forage kochia is broadly adapted to various semiarid rangelands (McArthur et al. 1996; Harrison et al. 2000), has high salt and alkali tolerance (Francois 1976), and is competitive against the annual noxious weeds cheatgrass (*Bromus tectorum* L.) and halogeton (*Halogeton glomeratus* [Stephen ex Bieb.] C.A. Mey.) (Stevens and McArthur 1990; Monaco et al. 2003). Newhall et al. (2004) reported that forage kochia was one of few species capable of establishing and competing with cheatgrass (*Bromus tectorum* L.) in a salt desert shrub environments frequently experiencing wildfires, severe wind erosion of topsoil, and drought. Monaco et al. (2003) conducted research in a similar environment that historically had been used for winter grazing of sheep, but where overgrazing and repeated wildfires had completely eliminated all perennial shrubs leaving only a monoculture of cheatgrass. They reported that forage kochia established, persisted, and reduced the biomass and frequency of cheatgrass during a 10 year period with annual precipitation ranging from 127 to 200 mm. Harrison et al. (2002) reviewed the characteristics that make forage kochia a good plant material for greenstrips to stop wildfires in semiarid regions. The study sites in the Newhall et al. (2004) and Monaco et al. (2003) papers have both successfully stopped wildfires since their establishment.

Nutritional and Grazing Value

The literature suggests that forage kochia is most abundant in the countries of Kazakhstan, Uzbekistan, and Kirghistan, where it is recognized as an important fall and winter forage for sheep, cattle, horses, camels, and wildlife (Balyan 1972; Waldron et al. 2001; Gintzburger et al. 2003; Waldron et al. 2005). Waldron et al. (2010a) reviewed forage kochia's nutritional attributes. In summary, forage kochia has high crude protein (CP) (> 70 g/kg) during the critical fall/winter grazing period (Davis 1979; Davis and Welch 1985; Waldron et al. 2006), low non-toxic levels of oxalates (Davis 1979), acceptable digestibility (Welch and Davis 1984; Davis and Welch 1985; Waldron et al. 2006), increased rate of fiber digestion (Waldron et al. 2010a), and high palatability to livestock (Waldron 2010b). Waldron et al. (2006)

recently documented that forage kochia maintained or improved body condition of cattle during the winter without any additional protein or nutrient supplementation and resulted in a 25 percent economic savings over alfalfa hay feeding practices.

Forage kochia's forage yield potential varies depending upon the subspecies and environment, but generally ranges from 1000 to 6000 kg/ha (Balyan 1972; Nechaeva 1985; Gintzburger et al. 2003; Waldron et al. 2006; Waldron et al. 2010a) and in almost all cases, the reported yields represent a 3 to 6 fold increase in forage production as compared to existing rangeland without forage kochia. In the western U.S., forage kochia is being used to provide critical forage and habitat for wildlife, including deer, antelope, wild horses, and birds. This value is most pronounced in areas where wildfires have destroyed native vegetation.

Collection and Breeding

Breeding of improved cultivars appears to have begun in the 1970s in the former U.S.S.R. resulting in several improved cultivars of forage kochia (Alimov and Amirkhanov 1980; Herbel et al. 1981; Nechaeva 1985; Rabbimov 1984); however, these are not commercially available at this time. Krylova (1988) and Harrison et al. (2000) independently reviewed the introduction, cultivar development, and cultivation of forage kochia in the U.S. In brief, forage kochia was introduced to the U.S. in 1966 by researchers looking for a plant to suppress halogeton on droughty and saline soils (Harrison et al. 2000). One germplasm accession was selected and released as the cultivar 'Immigrant' in 1984 based upon its overall persistence, forage production, forage quality, palatability, and competitiveness with annual weeds (Stevens et al. 1985). Immigrant remains the only released cultivar of forage kochia in the U.S., and is a short-statured, diploid, subspecies *virescens* type used for livestock and wildlife forage, soil stabilization, rangeland reclamation, and suppression of wildfires.

The USDA-ARS Forage and Range Research Laboratory in Logan, Utah currently has an active forage kochia research and breeding program. This program researches the use and establishment of forage kochia, and develops larger statured, more productive, easier establishing cultivars with a goal to

enhance its utilization as winter forage in temperate, desert regions. Scientists from this lab have led collection trips to Kazakhstan (Waldron et al. 2001) and Uzbekistan (Waldron et al. 2005) collecting over 250 accessions of forage kochia. Improved, taller statured breeding lines are being evaluated throughout the western U.S. and in the country Jordan (Bailey et al. 2010).

CONCLUSION

Kochia prostrata is a valuable forage plant for livestock and wildlife in the temperate, semiarid and arid regions of central Asia. In these areas, it is known as the “alfalfa of the desert” (Waldron et al. 2005). It is not as common in the semiarid western U.S., but it is often recommended for reclaiming degraded rangelands, in part because of its competitive advantage over cheatgrass, as well as its value as forage and habitat for livestock and wildlife. It is extremely drought and salt tolerant, often growing in extremely harsh environmental conditions that preclude the successful establishment of other plant species. Research and experience have shown that forage kochia is a very palatable and nutritious shrub, especially during the fall and winter when nutritional quality of other plants is low. Its nutritional characteristics include CP levels above the 70 g/kg needed for ruminant animals, acceptable fiber levels, and low tannins and oxalates. Because of its competitive ability, some are concerned about forage kochia becoming an invasive weedy species. However, research and long-term observations indicate that forage kochia competes well with annuals, but does not readily invade perennial plant communities. Forage kochia has the potential to improve the habitat and sustainability of wildlife populations in areas that are threatened with frequent wildfires, extended drought, and increasing salinity.

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Reinterpreting Historical Data for Evidence-Based Shrubland Management

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ABSTRACT

Long-term vegetation dynamics in the Chihuahuan Desert of southern New Mexico have been intensively studied for over a century, and interpretations of the broad scale drivers of these dynamics are numerous. We now understand that interpretation of spatially heterogeneous change requires a more nuanced, contextualized, and detailed understanding of edaphic features and landscape characteristics. Recently, state and transition models (STMs) have been employed to represent landscape-specific dynamics for each ecological site within a Major Land Resource Area (MLRA). We re-examined data characterizing vegetation across the public lands of the northern Chihuahuan Desert at two points in time, the 1930s and 2005. In this study, our objectives were to (1) develop geospatial data layers of historical and current vegetation states, (2) compare vegetation states between the 1930s and 2005 where the two data layers overlap, and (3) interpret any major vegetation state changes over this ~70 year period within the context of specific ecological sites. It was our hypothesis that ecological dynamics would vary in interpretable ways among ecological sites. Three primary observations are drawn from our results: (1) the bulk of the region was relatively stable during this period, (2) approximately the same amount of area experienced increased grass dominance as experienced increased shrub dominance, and (3) dynamics are strongly influenced by the properties of specific ecological sites. Major vegetation state changes, involving either increased grass dominance or increased shrub dominance, only occurred to any extent in 11 of 18 ecological sites within this study area. More important to management, significant increases in shrubs occurred within only four ecological sites. These sites were sandy, deep sand, shallow sandy, and gravelly sand. All other ecological sites within this region were relatively stable over the ~70 year period between observations. The obvious management implication is the importance of stratifying by ecological site prior to application of shrub control treatments.

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INTRODUCTION

Vegetation dynamics in the Chihuahuan Desert of southern New Mexico have been studied for over a century (Buffington and Herbel 1965; Gibbens and others 2005; Schlesinger and others 1990; Wootton 1908). These studies have produced a long-term record indicating significant and lasting vegetation change (Havstad and others 2006; Peters and others 2006). Though the interpretations of the broad scale drivers of these changes are numerous and diverse (Van Auken 2009; Yanoff and Muldavin 2008), ecologically-based principles with application to rangeland management have been drawn from these studies for decades (Herbel and Gibbens 1996; Jardine and Forsling 1922). Increasingly, though, we

have understood that interpreting land change requires a more detailed and location specific understanding of edaphic features and landscape characteristics that contribute to resistance and resilience of vegetation assemblages across this arid region (Bestelmeyer and others 2009).

Central to this improved approach to interpreting land change have been state and transition models (STMs), rooted in a thorough understanding of vegetation dynamics and linked to specific ecological sites and their descriptions (Bestelmeyer and others 2004). The Natural Resources Conservation Service (NRCS) has made recent advances in the development of rangeland ecological site descriptions

(ESDs) and the mapping of ecological sites, especially within Major Land Resource Area (MLRA) 42, which encompasses much of southern New Mexico (see: <http://www.cei.psu.edu/mlra>). The STMs embedded within these ESDs, when used either explicitly or implicitly, provide a mechanism to house and disseminate information including an understanding of current vegetation states (Bestelmeyer and others 2003), explanations of long-term dynamics (Yao and others 2006), and evaluations of management actions (Havstad and James 2010).

Techniques that utilize remotely-sensed imagery, including aerial photographs, to map vegetation states within this region are well established (Laliberte and others 2004). In fact, remotely-sensed imagery has been available since the 1930s in some areas for detection of vegetation states and recent dynamics (Browning and others 2009). In addition, detailed field observations of vegetation conditions have been available for this region since the establishment of the Grazing Service, the forerunner of the Bureau of Land Management (BLM), following passage of the Taylor Grazing Act by the US Congress in 1934 and subsequent establishment of public land grazing districts across the western US (Skaggs and others 2011, in press). Ground-based surveys conducted in conjunction with the establishment of public land livestock grazing districts in the 1930s provided systematic and geographically extensive records of historical vegetation conditions. These records can be extremely useful for tracking vegetation changes through time and placing these changes within the context of other relevant geospatial data.

Combining historical field data, remotely-sensed imagery, state and transition models, ecological site descriptions, and knowledge of broad scale drivers allows for spatially-explicit interpretations of vegetation dynamics across the region. Our objectives were to (1) develop geospatial data layers of historical and current vegetation states, (2) compare vegetation states between the 1930s and 2005 where the two data layers overlap, and (3) interpret any major vegetation state changes over this ~70 year period within the context of specific ecological sites. It was our hypothesis that patterns of state change would vary among ecological sites.

METHODS

Study Region

This study was mostly confined to public lands within MLRA 42 administered by the BLM. The specific area of study was a region of approximately 8000 km² (2 M surface acres) across six counties in southern New Mexico (figure 1). Land use within this region has been dominated by cattle ranching over the past 125 years. Although livestock numbers are greatly reduced from those recorded in the early part of the 20th Century, the BLM Las Cruces District Office currently manages 603 grazing allotments. The region is characteristic of the northern extent of the Chihuahuan Desert (Havstad and others 2006) with its arid climate (long-term average annual precipitation <250 mm primarily as convectional storms in the summer months) and elevations above 1100 m (3600 ft). The area considered for analysis was necessarily restricted to regions of overlap between historical and modern datasets. More specifically, the study area was defined by those historical 1930s-era map polygons more than 70 percent covered by our current vegetation state map (see below).

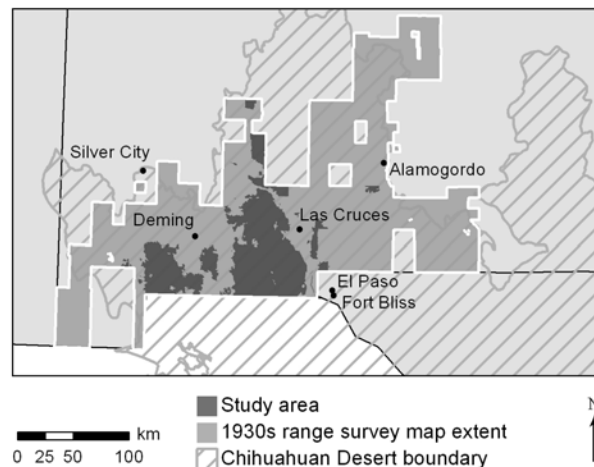


Figure 1. Extent of study area within southern New Mexico.

Current Ecological Site and State Mapping

In our approach to contemporary ecological site and state mapping, the basic stages are: (1) identify soil map units, (2) digitize vegetation states based on shrub cover/density and perennial grass cover/presence, and (3) attribute each polygon with

an ecological site and state. Ecological states were manually delineated in ArcGIS (Esri 2008) using color infrared, 1-m resolution 2005 Digital Ortho Quarter Quads (DOQQs), ground-based observations, and other geospatial reference layers. Soil Survey Geographic Database (SSURGO) soil map unit polygons were clipped to produce sub-polygons (child polygons) representing an ecological state or complex of ecological states based upon the state and transition model (STM) for the correlated ecological site (figure 2). Child polygons created in this manner differed from one another in the presence/absence or cover/density of perennial grasses and shrubs. Polygons were attributed with generic, three-digit state codes using ground-based spatial data, reference layers, photo-interpretation, and the associated ecological site description's STM. The dominant state was recorded as the first number in the state code. Where more than one state occurred within a polygon, the other two were recorded sequentially based on area. Otherwise, zeros followed the first (or second) number in the three-digit state code.

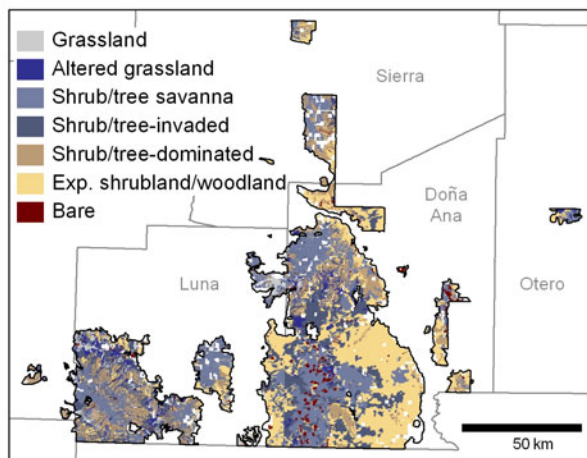


Figure 2. Dominant ecological states of the 2005 state map in regions of sufficient overlap (>70 percent) between historical and contemporary map polygons.

Reinterpretation of Historical Data

Detailed vegetation maps were produced in the 1930s by trained field personnel working for the Grazing Service. These maps, often referred to as "adjudication" or "range survey" maps because they indicated private and public land ownership boundaries, landscape features, and vegetation

related to newly established Department of Interior grazing districts, were based on ground observations directed by specific protocols. Skaggs and others (2011 in press) have detailed the procedures used to create the original 1930s maps and to convert the physical maps into a digital form for the portion of southern New Mexico studied here. Like the modern state map, the 1930s maps are object-based representations that segment the landscape into discrete vegetation polygons. Data recorded for each polygon include a list of up to five plant species. 1930s map polygons are, however, on average much larger (1392 hectares) than state map polygons (32 hectares) within our study area. Thus, two primary steps were taken to facilitate comparisons between the 1930s range survey maps and 2005 ecological state map.

These steps included (1) reclassification of map content to a compatible thematic format and (2) generalization of thematic information to a consistent spatial resolution. The modern state map provided the thematic template for the analysis, while the historical maps defined the spatial template. A rule set was developed to assign each 1930s map polygon to one of nine classes using the species listed for that polygon (table 1). This algorithm took into account the functional importance of different species and was meant to align the historical data as closely as possible with contemporary ecological state definitions. Five of the new classes developed for this study had a single equivalent class in the modern ecological state map. One new class, grass-dominated, included both shrub/tree savanna and shrub/tree invaded categories of the modern map, while three others had no counterpart in the modern classification scheme. In reclassifying the 1930s range survey maps, it was necessary to assume that plant species recorded for each polygon were the dominant species, listed in the order of their dominance, and that the protocol for recording species was regionally consistent. These assumptions appear reasonable given range survey methods of the time (USDA 1940). Nevertheless, a small change in species ordering could mean assignment of a polygon to a different generalized state (table 1). While up to three classes are recorded for each polygon of the modern state map, these polygons were reclassified to the new format using only that state indicated as dominant.

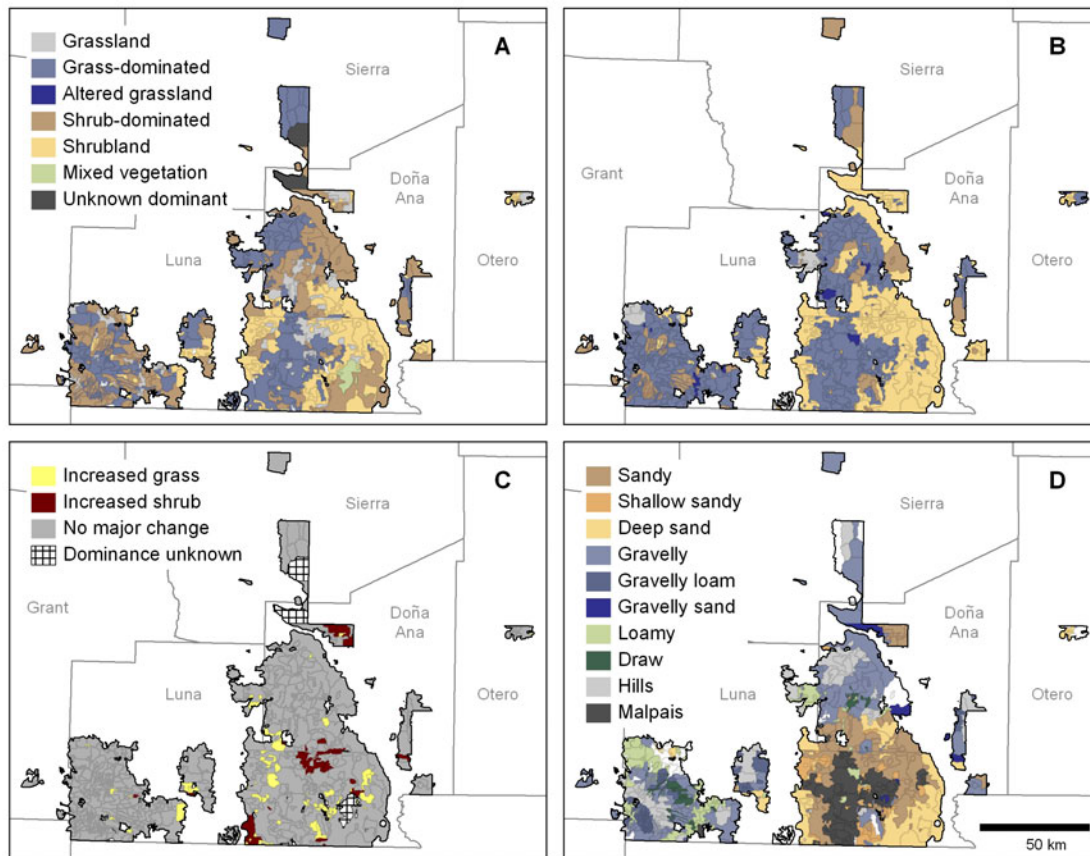


Figure 3. 1930s range survey map polygons classified by **A** generalized ecological state based on 1930s data, **B** generalized ecological state based on 2005 data, **C** major state changes (a departure of 2 or more generalized states) between the 1930s and 2005, and **D** dominant ecological site.

A second major step in facilitating comparisons between historical and modern datasets was to generalize the modern data to the coarser scale of the 1930s maps. This step was accomplished by merging the two datasets in a geographic information system (GIS) and calculating the area of each 1930s range survey map polygon intersected by contemporary ecological site and state classes. Each 1930s map polygon was subsequently assigned the generalized state and ecological site occupying the greatest proportion of the polygon (figure 3). A considerable amount of information was lost in the process. Yet, 1930s map data were interpreted as describing the predominant character of the landscape, and generalization of the state map was expected to produce a similar result. Grassland and altered grassland are not states recognized in the STMs of some of the ecological sites studied here, including deep sand, gravelly, gravelly loam, gravelly sand, hills, limestone hills, limy, and malpais. These ecological sites tend to feature scattered shrubs at

potential as described in current ecological site descriptions. Therefore, once 1930s map polygons were assigned a dominant ecological site, a final historical state classification was determined (table 1). Even if perennial grasses and no invasive shrubs were recorded for a particular 1930s map polygon, this polygon was classified as grass-dominated if it predominantly encompassed one of the ecological sites listed above, the presumption being that areas without shrubs were likely not at equilibrium and would eventually progress to a grass-dominated state, or that scattered shrubs might have been ignored by the recorder. This final step helped to further align the historical and modern classification schemes.

State changes between the 1930s and present were visualized by mapping the historical and modern states attributed to each 1930s map polygon (figure 3). The prevalence of different states was also examined by ecological site class for the two time periods. The percentage of an ecological site class

covered by each state was calculated using the equation

$$P = (\sum A_{\text{state}} / \sum A_{\text{ecological site}}) \times 100$$

where $\sum A_{\text{state}}$ is the area of all 1930s map polygons attributed with the generalized state and ecological site of interest and $\sum A_{\text{ecological site}}$ is the area of all

polygons attributed with the ecological site of interest. Because of the various assumptions, generalizations and considerable spatial data manipulations involved in this project, we focused on major vegetation changes and placed low confidence in interpretations involving ecological sites represented by few polygons (table 2).

Table 1. Rule set used to reclassify modern and historical maps.

1930s map species list ^a	Generalized state	2005 state map class ^b	Generalized state
2GRAM, ARDI5, ARIST, ARPU9, ARPUF, BOCU, BOER4, BOGR2, BOHI2, BOUTE, MUPO2, PLMU3, SCBR2, SPAI, SPCR, SPGI or SPORO listed first. FLCE, JUMO, LATR2, PRGL2, QUERC, QUTU2 or browse not listed. ATCA2 and ARFI may be present.	Grassland ^c , Grass-dominated ^d	Grassland	Grassland
2GRAM, ARDI5, ARIST, ARPU9, ARPUF, BOCU, BOER4, BOGR2, BOHI2, BOUTE, MUPO2, PLMU3, SCBR2, SPAI, SPCR, SPGI or SPORO listed first. FLCE, JUMO, LATR2, PRGL2, QUERC, QUTU2 and/or browse also listed.	Grass-dominated	Shrub/tree savanna or Shrub/tree-invaded	Grass-dominated
Perennial grass species other than those above listed first and not DAPU7.	Altered grassland ^c , Grass-dominated ^d	Altered grassland	Altered grassland
ARFI, ATCA2, FLCE, JUMO, LATR2, PRGL2, QUERC, QUTU2 or browse listed first. 2GRAM, ARDI5, ARIST, ARPU9, ARPUF, BOCU, BOER4, BOGR2, BOHI2, BOUTE, PLMU3, SCBR2 and/or SPAI also listed.	Shrub-dominated	Shrub/tree-dominated	Shrub-dominated
ARFI, ATCA2, FLCE, JUMO, LATR2, PRGL2, QUERC, QUTU2 or browse listed first. 2GRAM, ARDI5, ARIST, ARPU9, ARPUF, BOCU, BOER4, BOGR2, BOHI2, BOUTE, PLMU3, SCBR2 or SPAI not listed. MUPO2, SPCR, SPGI and SPORO may be present.	Shrubland	Expansion shrubland/woodland	Shrubland
Vegetation number 8 and no species listed.	Bare	Bare/annuals	Bare
Assemblage of shrubs, grasses and/or succulents not representing one of the above classes.	Mixed vegetation	N/A	N/A
The code listed first could not be translated to a modern species code.	Unknown dominant	N/A	N/A
Areas are delineated on the map but not surveyed.	Undefined	N/A	N/A

^aPlant species recorded for each historical map polygon were assumed to be the dominant species, listed in the order of their dominance. Species are referenced here by their current USDA plant symbol. In assessing species order, we ignored those shrub and succulent species not specifically referenced by their symbol in one of the above rule descriptions. Polygons whose species list included only these “not functionally important” shrub or succulent species were assigned the mixed vegetation class.

^bThe 2005 ecological state map was reclassified based on the dominant state within each polygon.

^cClass assigned to polygons dominated by bottomland, clayey, draw, loamy, loamy-gypsum upland-gypsum, salt flats, salty bottomland, sandy, or shallow sandy ecological site.

^dClass assigned to polygons dominated by deep sand, gravelly, gravelly loam, gravelly sand, hills, limestone hills, limy, or malpais ecological site.

Table 2. Percentage of area covered by each generalized state in the 1930s and 2005, by ecological site.

Generalized state	1930s						2005					
	Sandy ecological sites			Gravelly ecological sites			Sandy ecological sites			Gravelly ecological sites		
	Shallow	Deep		Gravell	Gravell	Gravell	Shallow	Deep		Gravell	Gravell	Gravell
	Sandy	sandy	sand	y	y loam	y sand	Sandy	sandy	sand	y	y loam	y sand
Grassland ^a	20.1	31.5	N/A	N/A	N/A	N/A	0	2.1	N/A	N/A	N/A	N/A
Grass-dominated	13	0	6.8	19.8	23.7	18.7	31.3	43.4	1.3	38.6	63.4	9.1
Altered grassland ^a	0	0	N/A	N/A	N/A	N/A	1.9	2.4	N/A	N/A	N/A	N/A
Shrub-dominated	9.8	0.6	50.2	58.8	64.4	69.6	0	0	0.3	33.5	30.7	0
Shrubland	50.7	66.7	42.7	12	10.1	11.7	66.8	52	98.4	27.9	5.9	90.9
Mixed vegetation ^b	6.3	1.2	0	0.1	0	0	N/A	N/A	N/A	N/A	N/A	N/A
Unknown dominant ^b	0.1	0	0	9.3	0	0	N/A	N/A	N/A	N/A	N/A	N/A
Undefined ^b	0	0	0.3	0	1.7	0	N/A	N/A	N/A	N/A	N/A	N/A
Total area (km ²)	1331	268	972	1877	328	121	1331	268	972	1877	328	121
Polygon count	83	37	36	111	42	9	83	37	36	111	42	9

^aGrassland and altered grassland are not considered stable states in the modern classification scheme for deep sand, gravelly, gravelly loam, and gravelly sand ecological sites.

^bThese states have no equivalent class in the 2005 classification scheme.

RESULTS AND DISCUSSION

State Changes 1930s to 2005

Our analyses worked from a fairly simple, but historically referenced model of vegetation states and transitions for this region. In general, the predominant ecological sites across the study area are characterized by five vegetation states: a grassland state dominated by historically dominant grass species (grassland), a grassland state dominated by grass species not considered to be historically dominant (altered grassland), a grass/shrub savanna (grass-dominated), a shrub-dominated state with some cover of historically dominant grass species and large unvegetated gaps (shrub-dominated), and a shrubland state lacking historically dominant grass species (shrubland). This generalized state model can be applied to nearly 95 percent of the study area (> 7500 km²) and at least 10 of the 18 main ecological sites within the region, including the area's six sandy and gravelly type ecological sites that are common across MLRA 42.

In characterizing vegetation dynamics using this generalized state model, we acknowledged two major differences between the historical and contemporary datasets: (1) differences in the spatial scale of the two state maps, and (2) differences in precision between

the modern state map attributed through photo interpretation, field observations and geospatial data layers and the historical state map derived from simple species lists recorded in the field. We thus focused on vegetation dynamics involving major state changes between the 1930s and 2005. These "major" changes were defined as a departure of two or more vegetation state classes over time based on our generalized state and transition model for the region. Considering, for example, a map polygon characterized as being predominantly grassland in the 1930s, a "major departure" would require that the polygon be characterized in 2005 as predominantly shrubland with some historically dominant grasses and large unvegetated areas (shrub-dominated), or predominantly a shrubland lacking historically dominant grasses (shrubland). If the polygon in 2005 was characterized as a grassland with shrubs present (grass-dominated), even though this designation might reflect a vegetation state less dominated by perennial grasses than in the 1930s, this would not be characterized as a major state change and would not be reflected in this analysis as having changed over the ~70 year period. The same required degree of departure would also apply to changes from shrubland or shrub-dominated states in the 1930s to grassland, altered grassland, or grass-dominated states in 2005. In applying this algorithm, altered grassland and grass-dominated states were given the same rank.

Based on these protocols, major state changes from the 1930s to 2005 are illustrated in figure 4. Three primary observations are drawn from these results: (1) the bulk of the region was relatively stable during this period, (2) approximately the same amount of area experienced increased grass dominance as experienced increased shrub dominance, and (3) dynamics differ strongly among ecological sites. To a great extent, these observations are counter to conventional interpretations of vegetation dynamics for this region drawn from anecdotal data. First, following the droughts of the 1930s and 1950s, it is typically assumed that major state changes occurred widely across the region. Second, it is usually assumed that most state changes were an increasing dominance of shrubs and that there has been a widespread loss of perennial grasslands. Third, it is generally assumed that these changes have occurred rather uniformly across diverse ecological sites.

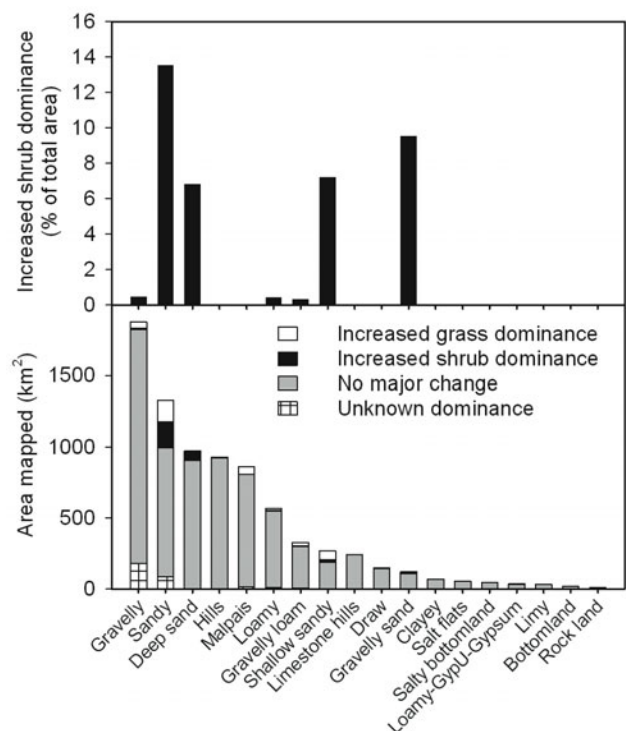


Figure 4. Major vegetation state changes (a departure of 2 or more generalized states) between the 1930s and 2005 delineated by the 18 major ecological sites within the study area.

It is certainly possible that much of the area shown as stable from the 1930s to 2005 in figure 3 actually experienced substantial vegetation state changes prior to the 1930s. Pre-1930s pressures, such as overgrazing by livestock and lengthy drought periods in the late 19th and early 20th centuries, are well

documented (Havstad and others 2006). However, the distribution of vegetation states in the 1930s, when stratified by ecological site, indicate site heterogeneity in resistance and resilience to disturbance factors, and it would be inappropriate to assume prior broad scale disturbances had resulted in uniform and widespread vegetation state changes, or that those changes would reflect universal degradation (figure 3). One conclusion that we can draw from these historical and quantitative perspectives is that there has been considerable spatial heterogeneity in response to broad scale drivers, such as regional multi-year droughts.

Vegetation Dynamics of Sandy Soil Textured Ecological Sites

There are three ecological sites characterized by sandy textured soils within MLRA 42 – deep sand (ref #R042XB011NM), sandy (ref #R042XB012NM), and shallow sandy (ref #R042XB015NM). These ecological sites are common across the northern Chihuahuan Desert, occupying nearly 15 percent of the region and about 30 percent of the area studied here. Our results for vegetation dynamics across these three sandy type ecological sites are presented in figure 5.

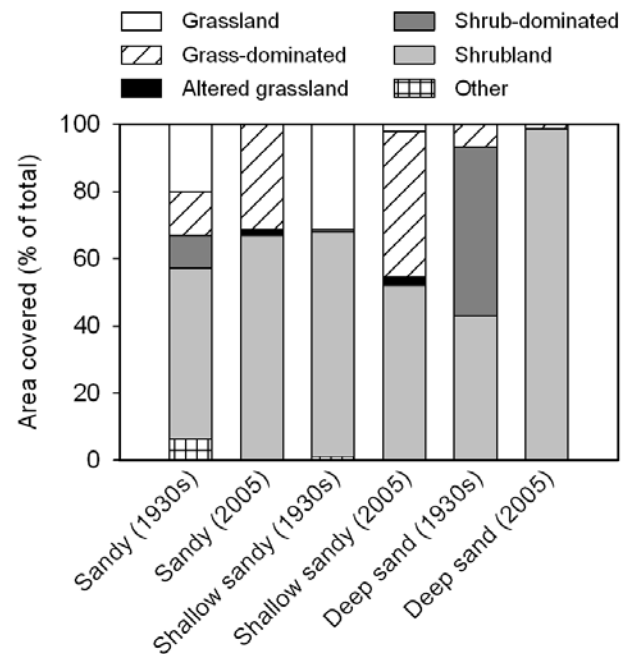


Figure 5. Vegetation state changes between the 1930s and 2005 delineated by the three principal sandy soil type ecological sites within the study area.

The deep sand ecological site was predominantly in either a shrub-dominated or shrubland state by the 1930s. By 2005 this ecological site was almost completely in a shrubland state across the study area. We conclude that this ecological site has poor resistance to extended drought, a conclusion recently supported by quantitative measures of relatively low plant available water in deep sandy soils lacking a calcium carbonate-cemented layer near the soil surface (Duniway and others 2010). This would support the observation of extensive shrubland and shrub-dominated states present across this ecological site prior to the 1930s. In addition, the poor resilience of this ecological site attributed to poor soil water retention features would help explain a near complete lack of the grass-dominated state in 2005 despite the implementation of various management practices, including more conservative livestock stocking rates, across this region since the 1930s.

Conversely, both shallow sandy and sandy ecological sites frequently exhibited grassland or grass-dominated states both in the 1930s and in 2005. The relative proximity of a calcium carbonate-cemented layer and/or a clay rich argillic horizon near the soil surface contributes to relatively high plant available water later within the growing season (McAuliffe 1994; Duniway and others 2010), and is likely one contributing factor to the resistance and resilience exhibited by these two ecological sites. However, it should also be noted that a large percentage of these sites were in the shrubland state by the 1930s, and these states appear to have been fairly stable for the ensuing ~70 years.

Vegetation Dynamics of Gravelly Soil Ecological Sites

There are three ecological sites characterized by gravelly surface textured soils within MLRA 42 – gravelly (ref #R042XB010NM), gravelly loam (ref #R042XB035NM), and gravelly sand (ref #R042XB024NM). Like the sandy textured ecological site group, these three ecological sites are fairly common across the northern Chihuahuan Desert, occupying nearly 20 percent of the region and about 30 percent of the area studied here. Our results for the vegetation dynamics across these three gravelly type ecological sites are presented in figure 6.

The gravelly sand ecological site has exhibited dynamics similar to the deep sand ecological site within this study area in MLRA 42. Vegetation states

across this site were predominantly either shrub-dominated or shrubland in the 1930s, and by 2005 most states were shrubland. Conversely, both the gravelly and the gravelly loam ecological sites exhibited an increase in the grass-dominated state from the 1930s to 2005. Although shrubland states are thought to be very stable, (Havstad and others 1999), we uncovered evidence of substantial grass recovery. These dynamics could be attributed to a combination of factors, including implementation of management practices such as shrub control, or the occurrence of climatic events that promoted successful grass regeneration. Our approach to reclassifying the 1930s map may also give the impression of state changes where no real changes have occurred, since small differences in the ordering of plant species listed for a polygon could mean the difference between a grass-dominated or shrub-dominated classification. It is also possible that map producers in the 1930s and 2005 used somewhat different parameters for defining species dominance. Because of these uncertainties in how the historical data were created, the line separating the grass-dominated and shrub-dominated states is likely less well defined than those separating other pairs of classes. The opportunity exists to further examine responses of specific areas within these sites to historical landscape treatments where records of treatment and response are available.

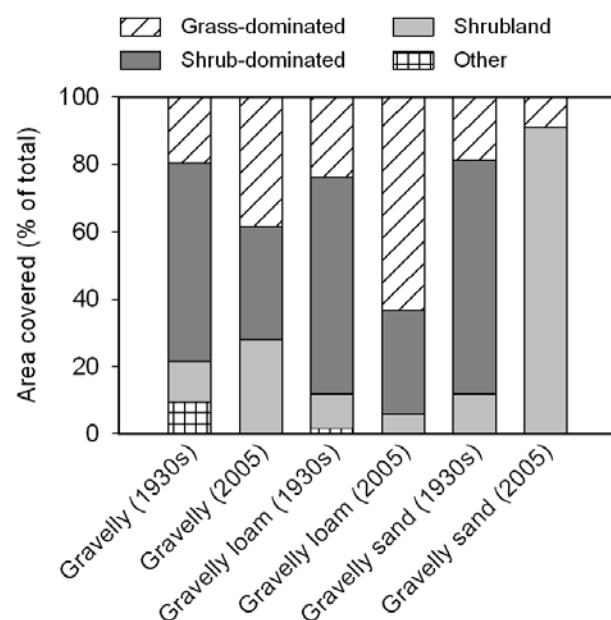


Figure 6. Vegetation state changes between the 1930s and 2005 delineated by the three principal gravelly type ecological sites within the study area.

MANAGEMENT IMPLICATIONS

There are 18 principal ecological sites within the study region. Major state changes between the 1930s and 2005, involving either increased grass dominance or increased shrub dominance, occurred in 11 of these ecological sites. Significant increases in shrubs occurred in only four ecological sites (figure 4). These sites were sandy, deep sand, shallow sandy, and gravelly sand. All other ecological sites within the region were relatively stable over the ~70 year period between observations. Although more detailed, site-specific studies are needed to reinforce the conclusions of this broad scale analysis, one clear management implication is the importance of stratifying by ecological site for application of shrub control treatments and in prioritizing management interventions or monitoring.

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The Yellowstone Sage Belts 1958 to 2008: 50 Years of Change in the Big Sagebrush (*Artemisia tridentata*) Communities of Yellowstone National Park

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ABSTRACT

*In 1958, 13 belt transects were established within the ungulate winter range in the northern portion of Yellowstone National Park to study how shrub communities were affected by grazing from ungulate populations. Between 1958 and 2008, the belts have been measured and photographed by different researchers at least once per decade, which has resulted in a comprehensive 50 year time series of how these communities have responded to climatic change, herbivory, and natural disturbance. In this study, we compare the percent cover, seedling establishment, and plant survival in these communities at two points in time (1958 and 2008); and explore which factors – climatic, herbivory, or disturbance – were most influential to changes in canopy cover and number of seedlings after 50 years. The recovery of the big sagebrush community after the North Fork fire is also discussed. Herbivory has controlled tree growth on the shrub belts. Climate and lack of disturbance have resulted in an increase in big sagebrush (*Artemisia tridentata*) cover on many shrub belts inside and outside of exclosures. Invasive annual species have become important drivers of vegetation change at the lowest elevation site.*

In Monaco, T.A. et al. comps. 2011. Proceedings – Threats to Shrubland Ecosystem Integrity; 2010 May 18-20; Logan, UT. Natural Resources and Environmental Issues, Volume XVII. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan Utah, USA.

INTRODUCTION

In 1957, Yellowstone National Park (YNP) managers embarked on an experiment to examine how ungulate populations affected vegetation in the northern portion of the park where many migratory species like elk (*Cervus elaphus*), mule deer (*Odocoileus hemionus*), and bison (*Bison bison*) spend their winter months (Edwards, unpublished letter). At the time, controversies over whether ungulate populations were too high and whether the browse vegetation was being overgrazed had existed for decades. As early as the 1930s, researchers raised concerns about declines in big sagebrush species possibly being related to overgrazing by overabundant populations of pronghorn (*Antilocapra americana*) and elk (Rush 1932). In 1950, Kittams concluded that big sagebrush was declining in many areas of the park for a combination of reasons, including physical breakage by browsing ungulates in winter, absence of seed production, and excessive browsing by pronghorn and elk at lower elevations in the park (Kittams, unpublished paper). In 1957, there were approximately 5000 elk in the park, 550 bison, 200 mule deer, and 150 pronghorn (Yellowstone National Park 1997). YNP managers were severely criticized

for allowing the populations of several of the ungulate species, especially elk and bison, to increase to levels that were thought to be detrimental to their winter range habitat and forage even though a policy of permitted hunts and culling kept the elk and bison herds at unnaturally low populations throughout the 1950s and 1960s (National Research Council 2002). Ranchers, park administrators, range managers, and park visitors believed that the range was being overgrazed, but little scientific data existed to support this belief (Yellowstone National Park 2005). By 1957, the National Park Service was concerned enough about the vegetation, the management issues, and particularly the declines in sagebrush, to initiate research that would provide scientific data to inform the debate and the regulation of ungulate populations in the park.

STUDY SITES

The ungulate winter range at the heart of the ungulate-management controversy consists of approximately 550 mi² (140,000 ha) of grassland, shrubland, and forest that extends across the northern boundary of the park (figure 1). The species that seasonally occupy this area include bison, elk,

pronghorn antelope, Rocky Mountain bighorn sheep (*Ovis canadensis canadensis*), moose (*Alces alces*), and mule deer (Barmore Jr. 2003). Since the 1980s, mountain goats (*Oreamnos americanus*) have also occupied and utilized this area as winter range (Yellowstone National Park 1997).

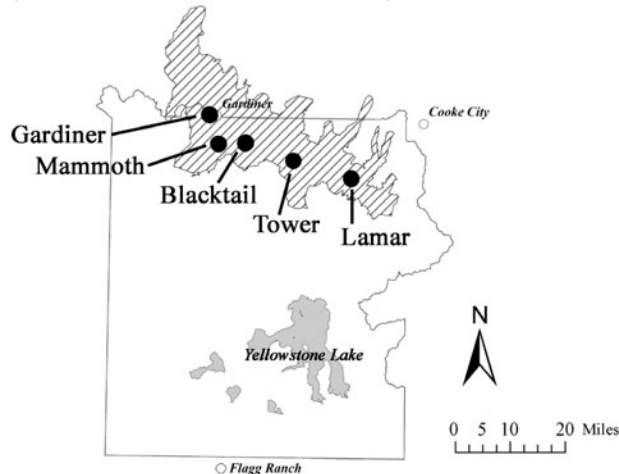


Figure 1. Boundary of the big-game winter range (striped area) and locations of the exclosures constructed in 1958 (dots). Winter range boundary provided by the Yellowstone Spatial Analysis Center.

In 1957, big-game exclosures were constructed at five locations across the northern winter-range area (figure 1). Park managers wanted to ensure that the study sites were designed and located so that they would provide for “detailed studies [for research] and demonstration areas to explain the wildlife range problem with the public” (Edwards, unpublished letter). The Tower exclosure was dismantled by 1962 because of a controversy over its visibility to the public, but the four remaining original exclosures still exist. In 1961, four additional exclosures were constructed to enhance the experimental design. They were constructed near the existing exclosures at Gardiner, Blacktail and Lamar and at a new location at Junction Butte. Because this study is focused on changes after a full 50 years, I do not address the changes that have occurred in the 1961 exclosures, although many of the same trends have occurred in them as in the 1957 exclosures.

Each ungulate exclosure constructed for the experiment is bounded by a fence over eight feet (2.4 m) tall that encloses approximately 5 acres (2 ha). The entire study design included two types of transects in both the original and 1961 exclosures – belt transects for mapping changes in sagebrush/browse, willow, and aspen; and line

transects (i.e., the Parker transects) for tracking changes in grasses, forbs, and shrubs; plus a square quadrat (9 ft²) for measuring percent cover changes in forbs and grasses. Two sagebrush belt transects were established at each location – one inside and one outside of the exclosure – for a total of eight belts to study changes in shrub cover in the 1957 exclosures (figure 2). Each belt transect was 5 ft (1.5 m) wide by 50 to 100-ft (15.2 to 30.5 m) long with the corners permanently marked with rebar. Originally the belt transects were called “sage belts” or “browse belts” depending on location. The dominant shrubs in the communities were, and still are, big sagebrush (*Artemisia tridentata*), green rabbitbrush (*Chrysothamnus viscidiflorus*), gray rabbitbrush (*Chrysothamnus nauseosus*), and horsebrush (*Tetradymia canescens*).

The line transects were established inside and outside of each exclosure in a paired configuration where slope, aspect, and elevation were matched as closely as possible between the inside and outside lines (figure 2). Each line transect was 100 ft (30.5 m) long and the total numbers of paired line transects vary with location. Because this study is focused solely on the shrub (particularly big sagebrush) changes over 50 years, the change in vegetation on the line transects will not be addressed in this paper except to put the design of YNP’s experiment and the fire effects after the 1988 fire in context. Photos could not be located for sage belts that were affected by the 1988 fires, so the nearest line transects are used to describe the fire-effects at the affected exclosure.

Together the eight sagebrush or browse belts presented in this analysis encompass a range of elevations, moisture conditions, soil depths, vegetation types, and disturbance effects and the unique characteristics of each study area enhance the overall study design. Similar sample sites were grouped by Singer and Renkin (1995) based on elevation, snowpack, precipitation, and big sagebrush species. Their characteristics include:

Low-elevation site: The Gardiner sage belts are the most northern sage-belt sites and are located near the town of Gardiner, Montana (figure 3). This area is the lowest in elevation (5400 ft; 1650 m) and driest of all the 1958 sage-belt sites (Barmore Jr. 2003). Precipitation averages 30 cm/yr (Singer and Renkin 1995). It is also within a spring and fall migration path for antelope (White 2009), and used by elk and mule

deer throughout the year (Houston 1982). Within the past five years, the area has been heavily invaded by non-native annual grass and forb species that currently affect total soil moisture and native-plant germination and growth in this part of the park (Hektner 2009). Wyoming big sagebrush (*Artemisia tridentata wyomingensis*) is the dominant sagebrush subspecies at the Gardiner site (Singer and Renkin 1995).

Mid-elevation site: The Mammoth sage belts are located in an area of active geothermal activity at Mammoth Hot Springs. They are less than 0.25 mi (0.4 km) from the hot springs, at an elevation of 6400 ft (1950 m), and situated within open areas of coniferous forests. Non-native species occupy the area, but most are perennials or grasses located along horseback riding and hiking trails.

High-elevation sites: The Blacktail sage belts are at approximately 6700 ft in elevation in rolling terrain between wetlands (below) and coniferous forest (above). They receive an average of 55 cm/yr precipitation. They are adjacent to a popular hiking trail used by tourists for backcountry access and fishing, but tourists cannot access the sage belts inside the exclosures without permission. The Lamar sage belts are located along US Highway 212 near the Lamar River. They are in an area heavily used by bison during the summer months and by visitors who watch the bison and elk herds. The Lamar sage belts are at 6700 to 6800 ft (2050 to 2070 m) in elevation, and they exist on steeper hillsides than any of the sage belts. They receive an average of 55 cm/yr of precipitation. The dominant big sagebrush subspecies at these sites are mountain big sagebrush (*Artemisia tridentata vaseyana*) and basin big sagebrush (*Artemisia tridentata tridentata*) (Singer and Renkin 1995).

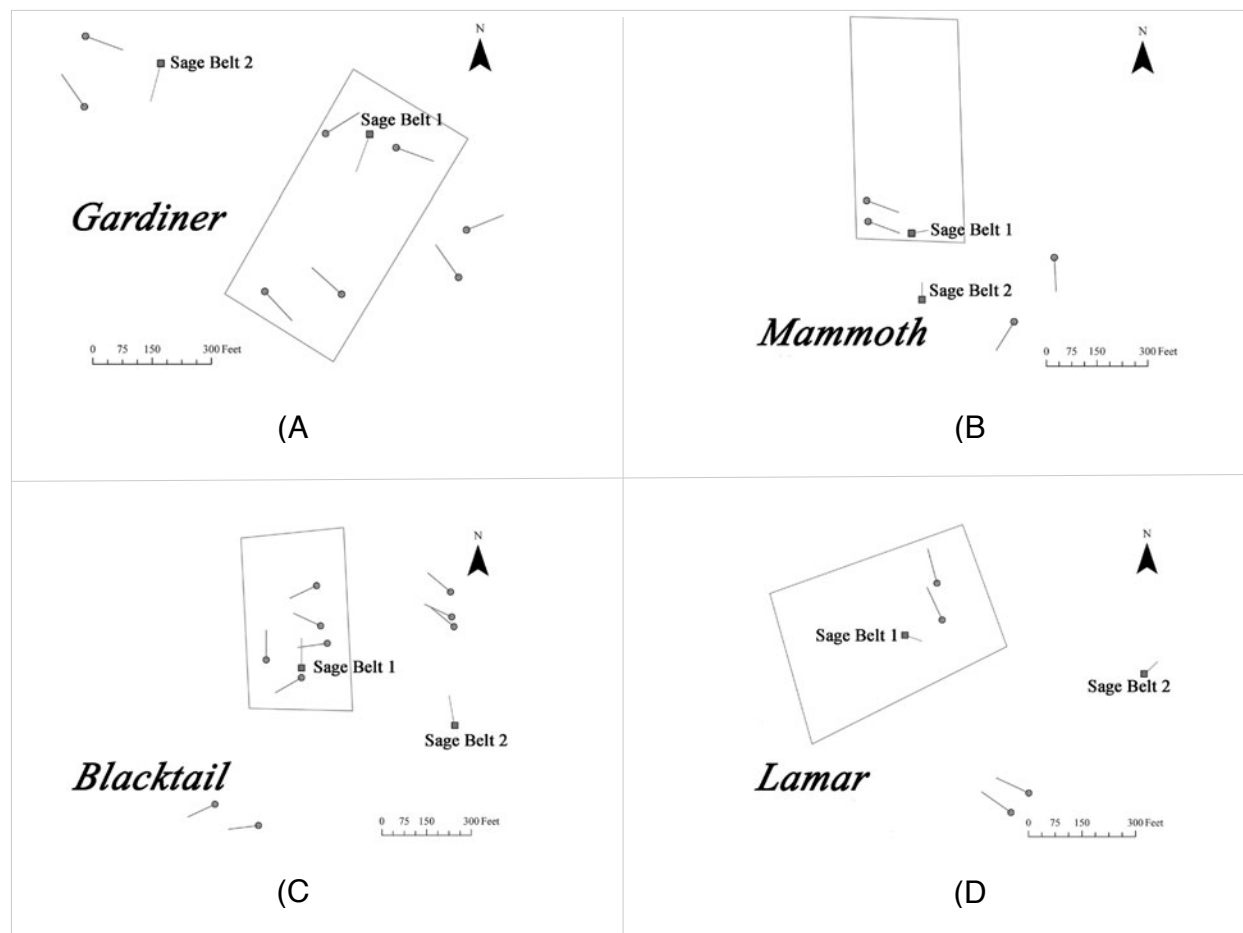


Figure 2. Locations of sagebrush belts (squares) and line transects (dots) within the (A) Gardiner, (B) Mammoth, (C) Blacktail, and (D) Lamar 1958 exclosures (outlined) that comprise part of Yellowstone National Park's natural experiment design. Sage-belt transects are labeled with their names. Bearing of each belt transect and line transect is indicated by directional lines.

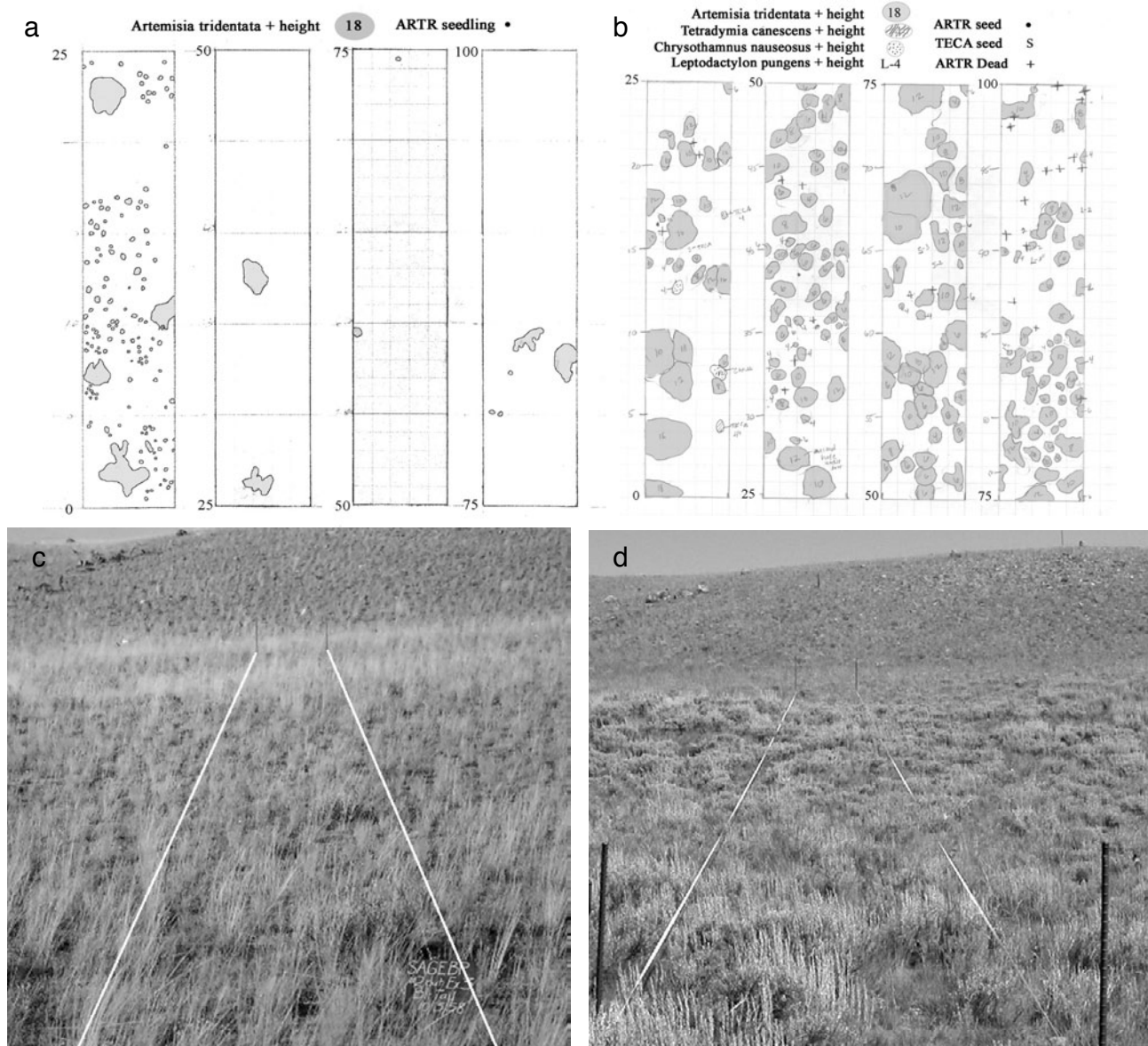


Figure 3. Data collection method from the Blacktail Sage Belt #2 (outside enclosure). (A) 1958 data form showing mapping of the aerial extent of sagebrush, species present, height of plants, seedlings, and dead shrubs (line in feet; tape location digitally enhanced); (b) 2008 data form showing mapping of the aerial extent of sagebrush and other shrubs, species present, height of plants, seedlings, and dead shrubs; (c) 1958 photo of belt transect corresponding to 1958 sample form; and (d) 2008 photo corresponding to 2008 sample form. Historic photo and data by Denton and Kittams (1958); 2008 photo by Art Sikkink.

METHODS

Sampling

The belt transects, which include the sagebrush or browse belts, were first sampled in 1958 by Gail Denton (Botany and Bacteriology Dept., Montana State College) and W.J. Kittams (YNP biologist) (Denton, unpublished data; Denton and Kittams, unpublished data; Kittams and Denton, unpublished data). Sampling consisted of mapping the location of each plant and the extent of the crown canopy by

species. The heights and dead vs. alive plants, by species, were recorded (figure 3a). The location and height of all seedlings and all dead shrubs were also identified. A photo point was established at the beginning of each sage belt during the original sampling and a photo was taken as part of the sampling procedure. Between 1958 and 2008, the belt transects have been sampled six times in much the same way, although height and/or seedling data were not measured in some years. Photos have been taken at similar locations on the belt transect each time the sage belts have been resampled.

In 2008, the eight 1958 sage belts were revisited for the 50th anniversary of YNP's experiment. The sage belt transects were sampled in the same way that they were sampled in 1958; namely by mapping the aerial extent of each plant to scale on graph paper, recording the species and height of each live shrub, mapping all seedlings by species at their germination location, and recording the location of all dead plants (figure 3). Photos were taken as per the sampling procedure and they were used in this study to augment descriptions of vegetation change at each sample location.

Evaluating Effects Of Burning Sagebrush In The Sage Belt Transects

Only the YNP North Fork Fire in 1988 burned any of the exclosure sites in this study. Its effects on the big-game exclosures were outlined in an unpublished report filed with YNP in September 1989 (Harter 1989). The report stated that the Blacktail exclosures were the only exclosures affected by the 1988 fires and that all three of the sage belts at Blacktail burned (Harter 1989). After the fire, minimal data was collected from the sage belts because there was little vegetation to map; burn severity estimates were made for the general area. Seedling heights and total seedlings were recorded, but individual seedling locations were not mapped according to the historic sampling protocols (Harter 1989). Because neither the sage nor transect belt photos from the 1989 fire have yet been located in the YNP archives, the best evidence of how the North Fork fire affected the sage belts are the changes that occurred on one transect line (Blacktail 58 C2T2), which is located within 10 ft (3 m) of the beginning of the inside belt transect (figure 2c). This paper uses data and photographs from the line transect to show fire effects and sagebrush recovery from the burn pictorially.

Data Analysis

This study was a qualitative and pictorial assessment of change within Yellowstone's experiment. Both historical and 2008 to-scale drawings were analyzed by (1) counting the number of grid squares covered by each shrub (by species) to determine a total canopy coverage of each species and (2) counting the number of seedlings, by species, on each belt transect for the two sample years. Change was assessed using tabular data, non-parametric statistical comparisons, and photographic records. Changes in canopy coverage and seedling counts between 1958 and 2008 were assessed graphically,

and Wilcoxon paired-samples tests were used to test for significantly different values in canopy cover and seedling counts between the two years. Locations inside and outside the exclosures were calculated separately ($n=8$). Significant differences were assessed if p -values were <0.05 .

Climatic trends in maximum and minimum temperature ($^{\circ}\text{F}$) and precipitation (inches) at the exclosures were assessed using data from the Mammoth Hot Springs weather station, which has been collected since 1955. Missing observations were not adjusted in any way.

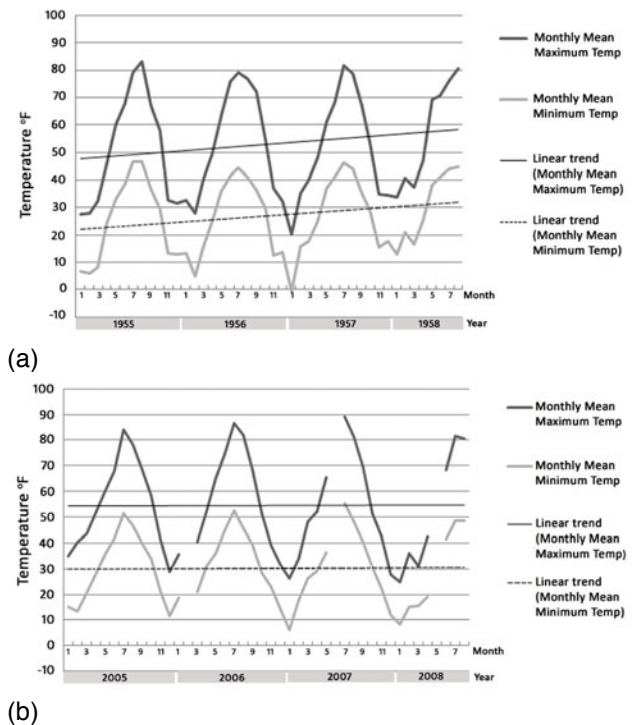


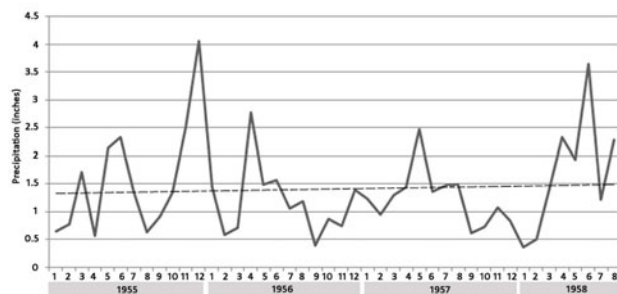
Figure 4. Monthly mean maximum and minimum temperatures in $^{\circ}\text{F}$ at Mammoth Hot Springs weather station in the four years prior to sampling the exclosure areas from (a) 01 Jan 1955 to 31 Aug 1958 and (b) 01 Jan 2005 to 31 Aug 2008. Linear trend lines are shown as solid and dashed lines.

RESULTS

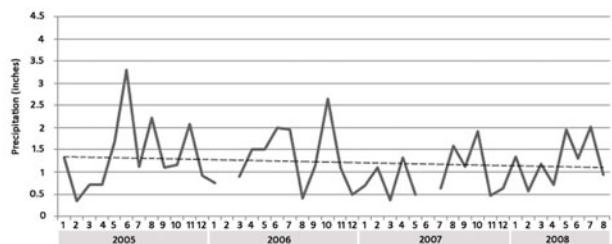
Climatic Trends (Mid-Elevation)

In the four years preceding 1958 and 2008, the park was experiencing different trends in temperature (figure 4a and 4b) and moisture conditions (figure 5a and 5b). The average annual temperature for the four-year period preceding sampling in 1958 was 39.8°F (4.3°C) and the average yearly precipitation was 16.44 in (41.8 cm) (National Climate Data Center

2010). The trend in mean monthly maximum and minimum temperature over that period was of gradually increasing temperatures and precipitation (figure 4a). The average annual temperature for the same period prior to sampling in 2008 was 41.33°F (5.2°C) and the average yearly precipitation was 14.25 in (36.2 cm). The trend in minimum and maximum monthly mean temperatures was flat while monthly precipitation declined each year (figure 5 b). The minimum and maximum temperatures in 2008 were at approximately 30 and 55°F (figure 4b), which were slightly higher than the mean minimum and maximum temperature in 1958 (figure 4a). In contrast to the spring and summer of 1958, which had an average of 2 inches (5 cm) rain each month before the initial sampling at the exclosures took place (figure 5a), the monthly precipitation in the spring and summer months before sampling in 2008 averaged approximately 1 in (2.54 cm). In general, the same trends that existed in 1958 continued at the mid-elevation weather station through 1974.



(a)



(b)

Figure 5. Monthly precipitation in inches at Mammoth Hot Springs weather station in the four years prior to sampling the exclosure areas from (a) 01 Jan 1955 to 31 Aug 1958 and (b) 01 Jan 2005 to 31 Aug 2008. Month number and year are shown on x-axis. Linear trend is shown as dashed line.

Compositional Changes Within The Belt Transects

Shifts in vegetation from dominantly grass to sagebrush have occurred on both grazed and ungrazed and low- and high-elevation sagebrush belts during the past 50 years (figure 6). Six of the

eight belt transects showed increases in canopy cover of big sagebrush that ranged from 5 to 45 percent. The largest increase in canopy cover occurred on the outside sagebrush belt at Mammoth (figure 6). The smallest increases occurred in the Lamar area. The average increase on the six belt transects was 24 percent. The remaining two belts had decreases of <5 percent each. Statistically, the differences between the eight location-year pairs were significant (p -value = 0.04).

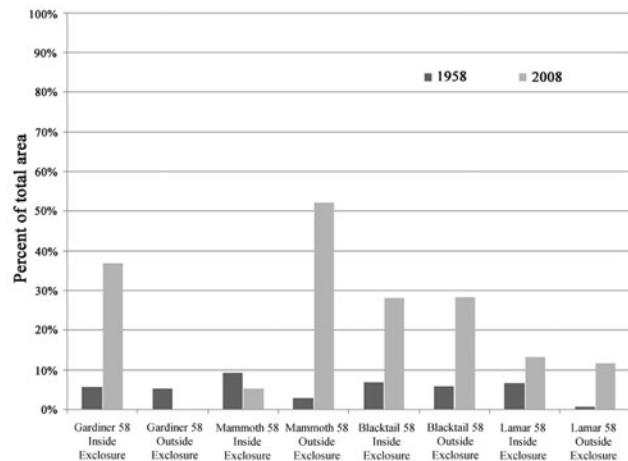


Figure 6. Differences in *Artemisia* cover between 1958 and 2008 for the eight sage belts established in 1957 and sampled in 1958.

The low-elevation, grazed site at Gardiner, the mid-elevation, ungrazed site at Mammoth (inside), and the high-elevation, ungrazed site at Lamar exhibited the most dramatic changes in composition over the 50 years. At Gardiner's outside sage belt, all shrubs that were part of the community for 30 years or more had died by 2008 (figure 7b). The native grass-*Artemisia* community that existed in 1958 (figure 7a) was replaced almost completely by short, non-native annuals, including annual wheatgrass (*Agropyron triticeum*), desert alysium (*Alyssum desertorum*), Japanese brome (*Bromus japonicas*), and cheatgrass (*Bromus tectorum*) (figure 7b). The same changes did not occur inside the Gardiner exclosure, where big sagebrush cover increased by 31 percent between 1958 and 2008. Inside the Mammoth exclosure, vegetation composition and structure changed from an *Artemisia tridentata*-dominated, open canopy community (figure 8a) to a community dominated by *Juniperus* and *Pseudotsuga menziesii* with an understory of *Symphoricarpos* (figure 8b). Conifers covered 30 percent of the belt in the Mammoth

exclosure after 50 years. On its paired belt outside of the exclosure, which contained less than 3 percent percent *Artemisia tridentata* in 1958 (figure 9a), sagebrush increased to 52 percent total cover by 2008 (figure 9b) and conifers occupied 0.01 percent of the belt.



(a)



(b)

Figure 7. Sagebrush reduction in Gardiner Sage Belt #2 (outside the exclosure). (A) In 1958, the belt contained mostly native grasses and sagebrush (YNP archive photo); (B) by 2008, the native grasses were gone, the sagebrush was dead, and the native community had been replaced by several non-native, annual forbs and grasses (Art Sikkink photo). Belt 100 ft (33.3 m) ends are marked with arrows in both photos.

The Lamar sage belts follow similar trends as those at Mammoth. Early photos of the inside belt transect show mostly grass and minor big sagebrush (figure 10a). All of the species that were identified on the inside belt transect in 1958 (Denton and Kittams, unpublished data) were still present in 2008, but aspen (*Populus tremenoides*), chokecherry (*Prunus*

virginiana), and service berry (*Amelanchier alnifolia*) had expanded to cover approximately 25 percent of the inside belt. Rose (*Rosa* sp.), snowberry (*Symphoricarpos* sp.), horsebrush (*Tetradymia canescens*), and green rabbitbrush (*Chrysothamnus viscidiflorus*) were also common shrubs inside the exclosure (figure 10b). In contrast, the belt transect that was open to grazing at Lamar had the same types of shrub species that were mapped in 1958, but all were too small to show on the photograph (figure 11a). All (except big sagebrush) were less than 4 in (10 cm) tall and presumably kept short by grazing. By 2008, canopy cover of big sagebrush had expanded to cover over 10 percent of the outside belt area (figure 11b).



(a)



(b)

Figure 8. Tree invasion in Mammoth Sage Belt #1 (inside exclosure) after 50 years of protection from grazing. (A) Originally, the belt was a sparse sagebrush and grass community (YNP archive photo); (B) by 2008 it was dominated by conifers and snowberry and consisted of less than 10% sagebrush (Art Sikkink photo).

In the areas where tree invasion was not a factor, big sagebrush coverage expanded approximately the same amount both inside and outside of the exclosures (figure 6), indicating that herbivory was not negatively affecting big sagebrush canopy cover. On the inside sage belts at Mammoth and Lamar, tree encroachment effectively decreased the area available for shrub growth so big sagebrush coverage shows a decline in total coverage between 1958 and 2008 (figure 6). It had not yet been eliminated from either site by 2008.



(a)



(b)

Figure 9. Sagebrush expansion in Mammoth Sage Belt #2 (outside the exclosure). (A) In 1958, the belt was dominated by grass with small sage plants and many seedlings (YNP archive photo); (B) by 2008, the belt was filled with sagebrush and trees were encroaching on its northern edge (Art Sikkink photo).

Whether total shrub diversity changed between 1958 and 2008 was hard to evaluate because, for most of the sites, big sagebrush was the only shrub mapped in 1958. Other shrubs were mapped on the same belts nine years later in 1967, but in 1958 the focus

was on big sagebrush and how it was affected by grazing. The only exception was the data collected on the belts at Lamar. At Lamar, several species of shrubs and trees were mapped in 1958 and 2008 so comparisons of diversity between the two years were easily made. The data showed that diversity increased at this high elevation site on belts both inside and outside of the exclosure. In 1958, there were six species of shrubs mapped on the inside sage belt and seven mapped on the outside belt. By 2008, there were eight species of shrubs plus aspen seedlings on the inside belt; and eight species on the outside belt (Sikkink, unpublished data 2008b). The two additional species included Oregon grape and green rabbitbrush.



(a)



(b)

Figure 10. Tree invasion in Lamar Sage Belt #1 (inside exclosure) after 50 years of protection from grazing. (A) Originally, the belt consisted of small plants of sagebrush, serviceberry, rose, snowberry, horsebrush, and green rabbitbrush, which were mapped on the sample form but are not obvious in the photo (YNP archive photo); (B) by 2008 all of the original species have grown and expanded, aspen and chokecherry have invaded the plot, and sagebrush is restricted to the last 20 ft (6 m) of the belt (Art Sikkink photo).



(a)



(b)

Figure 11. Sagebrush expansion in Lamar Sage Belt #2 (outside of the enclosure). (A) In 1958, the belt contained small sagebrush, serviceberry, green rabbitbrush, rose, horsebrush, and chokecherry (YNP archive photo); (B) by 2008, the original species were still present, Oregon grape had established, and the sagebrush had expanded to over 10% of the area (Art Sikkink photo).

Trends In Seedling Survival on Sage Belt Transects

Seedling counts differ by sample year and elevation (figure 12). In 1958, seedlings were much more common in the low to mid elevations (Gardiner and Mammoth) than they were in 2008. The average loss in number of seedlings at these locations was 18. At the higher elevations (Blacktail and Lamar), the opposite trend occurred in that there were more seedlings in 2008 than in 1958 both inside and outside of the enclosures. The average increase in number of seedlings for these two areas was 8. The difference in seedling counts between years was not significant ($p\text{-value} = 0.55$).

Fire Effects on the Shrub Communities

The shrub and grass communities of the enclosure areas have been remarkably unaffected by fires during the past 50 years. None of the transect lines or sage belts had burned prior to the NorthFork Fire in 1988. In 1988, records show that only the Blacktail enclosures were affected by fire and the entire set of sage belt transects had burned. The most recent sampling prior to the fires was in 1981 (Rominger and Cassirer, unpublished data). At that time, the southwest corner of the enclosure and the hillside outside the southwest was filled with mature sage plants (figure 13a). By 1994, the same area inside the enclosure was occupied by tall grass and young sage plants with extensive new growth on the branches; outside the enclosure, sage was scarce on the hillside (figure 13b). By 2008, mature sage was again abundant inside the enclosure (figure 13c), but sage still had not recolonized much of the hillside. In comparison to transect C2T2, the inside sage belt shows the same structure and composition (figure 13d). From 1981 to 2008, sagebrush increased from 14 to 28 percent in coverage inside of the enclosure and from 10 percent to 28 percent cover on the outside sage belt; and by 2008, there was very little evidence that the Blacktail sage-belt communities had burned at all except for a few fire-scarred stems and/or elevated root crowns on the shrubs, which indicated that the duff around the base of the plant had burned (Sikkink, unpublished data 2008b).

When compared to the other high-elevation sites at Lamar, sagebrush cover increased at both sites in both the grazed and protected areas (figure 6). By 2008, the Blacktail site showed the greatest increase in canopy cover (average 21 percent vs. Lamar average 8 percent) even though both of its big sagebrush belt transects had burned.

DISCUSSION

During the 50 years of YNP's experiment, the sage belts inside and outside of the enclosures have provided data on the relationships between herbivory and big sagebrush growth in the park. Today, new factors, such as climate change, tree invasion, and invasive non-native annual species, are also becoming important factors for change in the vegetation communities of YNP's northern winter range. When YNP's experiment began, the belt transects consisted mainly of native grasses. Big

sagebrush comprised less than 10 percent of the total area on any transect, regardless of whether it was grazed or protected from grazing by the exclosures (figure 6). Today, big sagebrush occupies a substantial area in most of the belt transects in both grazed and protected areas. Fluctuations in big sagebrush canopy cover, numbers of big sagebrush plants, seed leaders, and seedling survival on these belt transects have all provided different perspectives on vegetation change in the northern winter range throughout the duration of the experiment. Each new study fuels ongoing controversies over whether ungulate herbivory is the source of change in the big sagebrush communities and whether the ungulate population exceeds the carrying capacity of the northern range.

In this study, the effect of grazing on the vegetation of northern winter range is less clear-cut than some previous studies. Where Wambolt and Sherwood (1999) concluded unequivocally that herbivory was responsible for declines in canopy cover at all exclosure areas, this study found that there was an increase in sagebrush canopy cover at all belt transects except for Gardiner's outside belt (a low-elevation site) and Mammoth's inside belt (a mid-elevation site). Only Gardiner had an obvious decrease in big sagebrush that could be attributed, at least in part, to herbivory because big sagebrush was flourishing inside the exclosure and not outside. The results of this study agree more with Singer and Renkin (1995) who also found that big sagebrush cover increased in low-elevation areas where big sagebrush was protected from grazing but canopy cover increased in both grazed and ungrazed belts at high elevations. At the mid-elevation site, canopy cover of big sagebrush decreased between 1958 and 2008 because a majority of the shrub belt became covered by conifer trees. This result conflicts with Baker (2006) who stated that "the invasion [of conifer species], like juniper and Douglas fir, into sagebrush areas are not due to fire exclusion but other factors (i.e., grazing)." The area of tree invasion at the mid-elevation site is within the exclosure and well protected by grazing.

Differences in canopy cover between grazed and protected areas that were found in this study and those that were reported by Wambolt and Sherwood

(1999) can be explained in a number of ways. First, data on canopy cover were collected using different sampling methods. Data for this study were collected within the original, permanently-marked sage belt transect using historic mapping techniques. Wambolt and Sherwood (1999) sampled lines not associated with the original belt transects and used a line intercept method to determine canopy cover. Whereas this study focused solely on big sagebrush, Wambolt and Sherwood (1999) included both big sagebrush and other shrub species in some analyses. They also sampled both 1957 and 1962 exclosure areas, except for the burned areas at Blacktail, and included data from all of the areas in their statistical analyses. Singer and Renkin (1995) used methods comparable to the methods used in this study for their canopy cover results, but used circular plots inside and outside of six exclosures for utilization rates, biomass production, recruitment, and consumption. Differences in results and interpretations for all of the studies can also be attributed to the time frames that were sampled. Wambolt and Sherwood (1999) focused on herbivory and differences in vegetation cover within a single time frame. Singer and Renkin (1995) and this study compared differences between two points in time. This study concentrated only on the differences between the original data and new data collected in 2008, whereas Singer and Renkin (1995) included data from the 1960s and 1980s. If data from other sample years were included in the analysis for this study, interpretations would likely be different because some features, such as number of seedlings, have varied more over time.

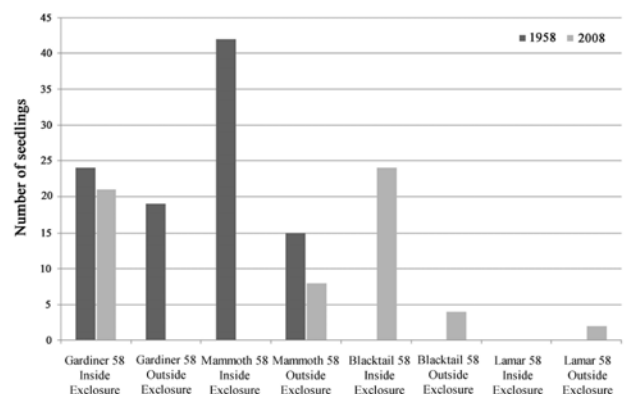


Figure 12. *Artemisia tridentata* seedlings present in the sage belts in 1958 and 2008. Differences in seedling counts between years are not statistically significant using a paired-sample Wilcoxon test (p -value=0.55).

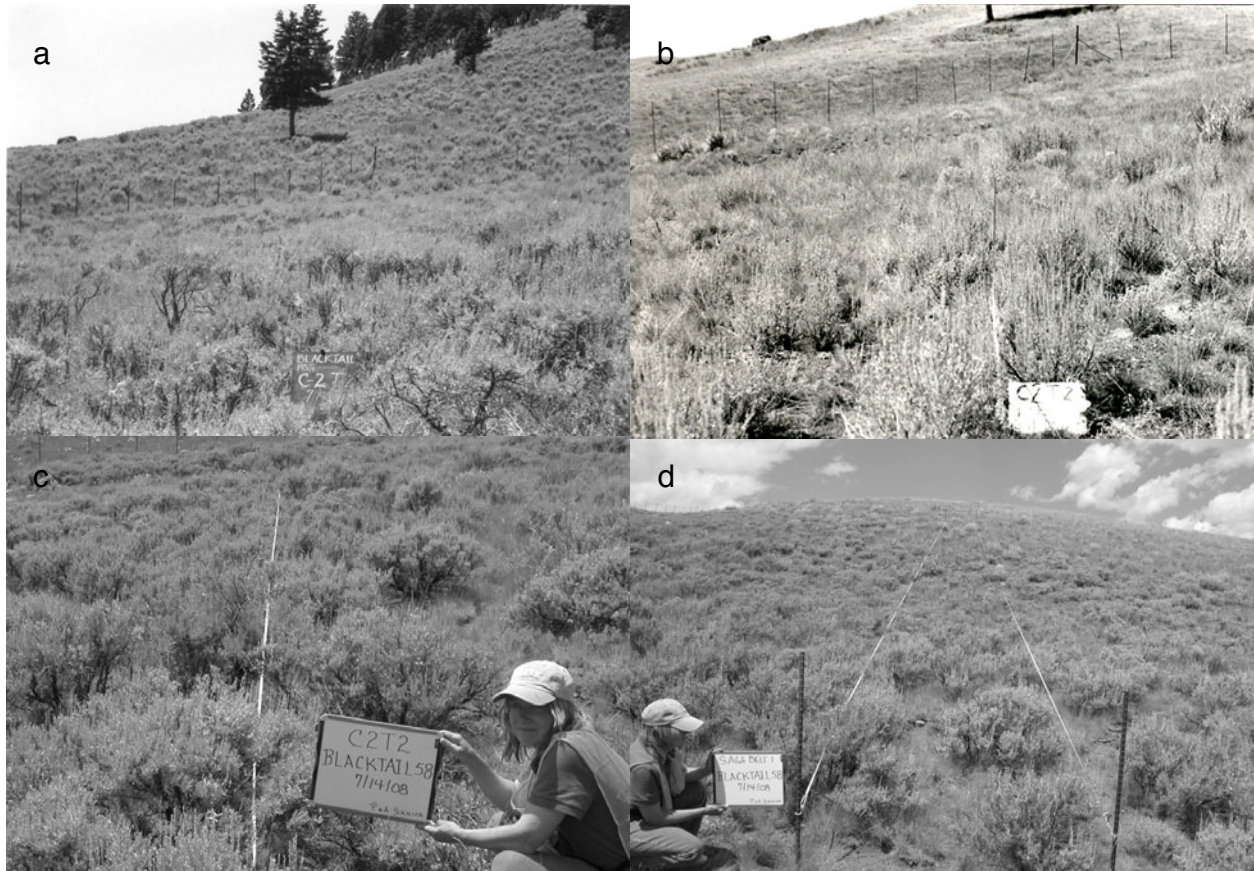


Figure 13. Effects of the 1988 North Fork Fire on *Artemisia* inside and outside of the Blacktail 58 enclosure. (A) Transect line C2T2 in 1981, prior to the North Fork Fire; (B) young sage regrowth on C2T2 six years after the burn in 1994; (C) mature sage on C2T2 in 2008; (D) sage growth on the sage belt adjacent to transect C2T2 in 2008. 1981 and 1994 photos from the YNP archive collection; 2008 photos by Art Sikkink.

In 2008, herbivory was still a dominant factor driving vegetation change at the low-elevation site in Gardiner, which agrees with several other studies (Houston 1982; Kittams, unpublished paper; Singer and Renkin 1995; Wambolt and Sherwood 1999). Inside the Gardiner enclosure, big sagebrush was flourishing in 2008 (figure 6), native shrub seedlings were relatively abundant (figure 12), and native grasses and forbs were present in amounts similar to those in 1958. Outside of the Gardiner enclosure, however, all shrubs had died, seedlings were non-existent, and non-native annuals had replaced most native grasses and forbs. The dramatic differences in shrub canopy cover and seedling establishment between the grazed and ungrazed areas leave little doubt that herbivory is very important in the area but it is not the only factor. Herbivory may be interacting with other factors to accelerate community change. Winter moisture for germination and warm, dry conditions during summer for growth create a favorable environment for growth of the annual non-native species, such as annual wheatgrass, brome,

and alyssum. These species have blanketed the landscape outside of the Gardiner enclosures in the past five to six years and affected soil moisture for growth and germination of the native species (Hektner 2009). How the declines in native species can be mitigated in the future is the subject of several new studies on restoration by the park that are occurring in the Gardiner area (Hektner 2009).

From the beginning of YNP's experiment to the mapping of the sagebrush belts in 2008, precipitation, temperatures, and grazing factors have changed dramatically. In 1958, the enclosure areas were sampled during a period of higher precipitation and cooler maximum and minimum mean temperatures than in 2008. Grass dominated all of the belts, and shrubs were kept small by grazing at the high elevations (indicated in the initial maps at Blacktail and Lamar). Ungulate populations were much smaller in 1958 than in 2008 because they were repressed by big-game hunting and culling within the park during the 1950s and 1960s (Singer and Renkin 1995;

Yellowstone National Park 1997). After four years of declining precipitation and higher temperatures preceding 2008, the high elevation sites showed the same percentages of sagebrush cover inside and outside of the exclosures, even though ungulate populations have increased significantly during the same time period. If herbivory alone were controlling the canopy cover of big sagebrush, cover in the outside belts should be much less than the inside belts, which is not the case for any of the mid- to high-elevation sites. Therefore, other factors besides herbivory must be contributing to the increase in big sagebrush cover. Climate and/or lack of disturbance are possible interrelated factors to explain these increases.

At mid to high elevations, herbivory and climatic effects are also important to controlling the growth and proliferation of trees. Conifers, service berry, and chokecherry, all regenerated and expanded in canopy cover when protected from herbivory by the exclosures. Similarly, data from willow and aspen belts inside and outside of the exclosures show that willows and aspens were able to grow to maturity inside of the exclosures, but they only existed as seedlings outside of the protected areas (Sikkink, unpublished data 2008a, c). Therefore, herbivory has been important to tree growth outside the exclosures at mid and high elevations as suggested by Wambolt and Sherwood (1999) and Kay (1995). However, mortality of willow and aspen trees has also increased inside the exclosures with the drier and warmer conditions of recent years (Bilyeu et al. 2008; Rogers 2008; Sikkink, unpublished data 2008a, c), suggesting that interactions between climate factors and herbivory affect growth and expansion at these elevations just like at Gardiner.

The shrub and grass communities of the exclosure areas have been remarkably unaffected by disturbance agents like fire during the past 50 years, but the belt transects at the Blacktail exclosures show how these high-elevation sites recovered from the North Fork fire in 1988. Six years after the sage belts burned, new plants and seed leaders were evident (figure 14b). By 20 years post fire, big sagebrush had surpassed its pre-fire canopy coverage percentages on both the inside and outside belt transects. The results from the Blacktail belts suggest that climate has controlled the recovery process of big sagebrush in the Blacktail area more than herbivory because the canopy coverage percentages are similar inside and

outside of the exclosure. The speed of recovery at Blacktail is remarkable in light of other studies that have followed the recovery of big sagebrush areas after burning (Cooper et al. 2007; Wambolt et al. 1998; Wambolt et al. 2001). Wambolt (1998) found that areas of Wyoming big sagebrush in the Gardiner Basin, which burned in 1974, recovered very little in 19 years. Welch and Criddle (2003) found that mountain big sagebrush recovered to 70 percent of pre-burn cover within 35 years. Colket (2003) showed that Wyoming big sagebrush in southeastern Idaho took 53 to 92 years to fully recover. Baker (2006) estimated even longer recovery rates of 50 to 450 years depending on big sagebrush type. Other studies have shown that big sagebrush recovery from burning is accelerated by dispersal of seed from nearby plants (Longland and Bateman 2002; Wroblewski 1999) or with soil seed pool immediately following a fire (Sugihara et al. 2006). Unlike the burn at Gardiner basin, the North Fork fire at Blacktail occurred in mountain and basin big sagebrush at high elevation where environmental and soil conditions were vastly different and seed sources were available from nearby areas.

Although most of the sites currently have abundant shrub cover and many mature shrub plants, the future of the shrub communities in YNP rests in production of seed to produce new plants, seedling survival, and maintaining community diversity. In 1958, when temperatures were cooler and precipitation more abundant, seedlings were more common at low to mid elevations. In 2008, under different climatic conditions, seedlings were much more common at the mid- and high elevation sites than they were at low elevation (figure 10), although the differences between the two years were not statistically significant at any location. The effect of grazing on big sagebrush seedling numbers is also not clear cut. In 2008, the low-elevation site at Gardiner showed seedling survival only inside the exclosure. The mid- to high-elevation sites at Mammoth and Lamar had seedlings only on the outside belts; but Blacktail had seedlings on both the inside and outside belts. It remains to be seen whether seed and seedlings will become more abundant with changes in climatic conditions. Evaluating changes in shrub diversity over time cannot be done using only the 1958 data because only big sagebrush was mapped in 1958. One thing is certain, this study does not show a decline in sagebrush canopy cover and the number of seedlings on most of the belt transects in YNP's

winter range as suggested by Wambolt (2005). All but two sage belt transects showed significant increases in big sagebrush canopy cover since 1958.

The controversies surrounding management of the northern range and whether it is being overgrazed or degraded over time will not be answered solely by the YNP sage belts because shrubs are not the preferred food for many ungulate species in the park (Singer and Renkin 1995). They do, however, constitute a portion of the diet of all ungulate species on the winter range (Singer and Renkin 1995; Wambolt 1998). Big sagebrush is a preferred food for pronghorn and mule deer, but not for elk or bison (Barmore Jr. 2003; Houston 1982; Singer and Renkin 1995). Even though big sagebrush comprises a small portion of an individual elk's diet, the numbers of elk on the northern winter range can have significant impacts on big sagebrush cover (Yellowstone National Park 1997). Individual transects, which provide data on grass, forb, and shrub composition inside and outside of the exclosures, will be more useful to evaluate changes in the grass and forb diversity and frequency that are most important for ungulate forage. In fact, Houston (1982), Coughenour et al. (1991) and Reardon (1996) have all addressed rangeland change using these transects and reported that the effects of herbivory were not significant on the exclosure sites between 1958 and the 1990s. This study focused on the trends in big sagebrush because that was the focus when the experiment was initiated in 1958. To determine the effect of herbivory on other shrub species in the northern range, the study must be expanded to include data for the years between 1958 and 2008 when mapping of the belts included other shrub species.

After 50 years, the sage belts indicate that climate (moisture and temperature), lack of fire, and tree invasion are major factors influencing change in these sagebrush communities. The results also suggest several interesting questions on the effects of invasive species in the park: such as, are the non-native species at Gardiner significantly changing the growing conditions for the long-lived species or are they simply taking advantage of climatic, management (past and present), and disturbance factors that facilitate their growth? Alternately, are the invasive species now a competitive or physical force for change at low elevations because of their dominance at Gardiner? Finally, the results suggest that changes in seedling survival are occurring in the winter range

that could affect future regeneration of sage in some areas. Yellowstone's experiment has provided valuable insights into the drivers of vegetation change over the past 50 years. It will continue to be important to park researchers and managers as they attempt to sort out the effects of herbivory, climate change, invasive species, and changing fire regimes on Yellowstone's vegetation over the next 50 years.

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Native Annual Plant Response to Fire: an Examination of Invaded, 3 to 29 Year Old Burned Creosote Bush Scrub from the Western Colorado Desert

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ABSTRACT

Creosote bush scrub vegetation typically contains high diversity of native annual plants relative to shrubs, cacti, perennial herbaceous species, or other plant life forms. This vegetation type is also very susceptible to exotic, invasive annual plants, which promote fire by changing fuel properties. The impact of fire on most perennial species is severe but the impact on native annual plants is not well understood. We measured annual species composition in five sites that each contained paired burned and unburned stands in the western Colorado Desert, California. The burned stands at each site ranged in time since fire from 3 to 29 years ago. Annual plant cover, species richness, and soil chemical and physical properties were compared in the paired burned and unburned reference stands. Differences between paired stands at the time of each fire are assumed negligible since shrub cover across fuel breaks did not differ prior to each fire based on aerial photographs. Fires elevated soil pH but otherwise had little effect on other soil properties. In recently burned stands, invasive annual grass abundance increased while native annual plant cover and species richness decreased. However, in older burned stands, annual plant composition did not always differ between paired stands because invasive annual plant abundance was very high in both stands. Thus, while fires can have long-lasting negative impacts to perennial components of creosote bush scrub, invasive species can displace native annual plants regardless of whether or not a site burns, although fire disturbance appears to accelerate invasive plant dominance.

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INTRODUCTION

Desert annual plants contribute a large proportion of the plant diversity found in creosote bush scrub vegetation (Jennings 2001). Many desert annuals are attractive wildflowers that can carpet the desert floor when the appropriate conditions are met (Goodpasture and others 2004). Unfortunately, exotic annual plants such as grasses (*Bromus madritensis*, *Schismus barbatus*, and *Schismus arabicus*), red-stem filaree (*Erodium cicutarium*), and Sahara mustard (*Brassica tournefortii*) have invaded and displaced native annual plants in many parts of California's deserts (Minnich 2008). In localized areas, invasive grasses have been especially problematic for fueling wildfires (Brooks and Matchett 2006; Brooks and Minnich 2006). One such area is the western edge of the northern Colorado Desert and southern Mojave Desert in Riverside and San Bernardino Counties, California (Brooks and Esque 2002). Fire is thought to be relatively common here

due to elevated precipitation and anthropogenic nitrogen deposition that promote invasive grasses (Allen and others 2009; Rao and Allen 2010; Rao and others 2010), and also because of increased human ignitions due to its location within a major wildland urban interface (Brooks and Esque 2002; Brooks and Matchett 2006).

Previous studies investigating the response of desert annual plants to fire have reported increases in *Schismus* spp. (Cave and Patten 1984; Brown and Minnich 1986; Brooks 2002) and *E. cicutarium* (Brown and Minnich 1986; Brooks and Matchett 2003). Larger seeded invasive annual grasses, like *B. madritensis* ssp. *rubens*, usually decrease immediately following fire (Brooks 2002; Abella and others 2009), but may return to or exceed pre-fire levels of abundance after several years (Brooks 2002). Species-specific responses to fire by native annual plants have been reported, but due to large differences in species composition between study sites, generalizations are

difficult to draw. However, several studies have found an increase in *Plantago ovata* following fire (Brown and Minnich 1986; Cave and Patten 1984; Brooks 2002). Lastly, there is evidence that fire in creosote bush scrub reduces annual plant species richness (Brooks 2002; Steers and Allen, in press), although other types of desert shrublands found at higher elevations in the Mojave and Sonoran Deserts, can have increased native annual plant species richness following fire when fires eliminate *B. madritensis* and no other invasive species become dominant (Brooks and Matchett 2003; Abella and others 2009; Steers, unpublished data).

The goal of this study was to document the impact of fire on native and exotic annual plants by investigating a number of burns that ranged in time since fire. Specifically, we wanted to determine how fire impacts native annual plant diversity. We also wanted to document whether fire promotes invasive annual plant abundance. Annual vegetation was sampled in the field from five sites that had burned from three to almost thirty years since the time of sampling. Our hypotheses were that fire would promote exotic annual plants and decrease native annual species richness in sites of varying age since fire.

MATERIALS AND METHODS

Study Area

The study area was located on the western edge of the Coachella Valley in Riverside County, California (figure 1). This valley forms the extreme northwest portion of the Colorado Desert and transitions into the Mojave Desert to the north, and into cismontane vegetation of the California Floristic Province towards the west and south. The primary vegetation in this region is creosote bush scrub (CBS) on the valley bottoms. A rich post-1960 fire chronosequence exists for the CBS vegetation that dominates the valley and eastern reaches of the Banning Pass, and several burned sites have been investigated previously (O'Leary and Minnich 1981; Brown and Minnich 1986). Average precipitation in the city of Palm Spring to the east of the study area is 13.1 cm while to the west, at Cabazon, average precipitation is 39.9 cm (WRCC 2008). Creosote bush scrub reaches its western-most extent in the Banning Pass near the eastern border of Cabazon.

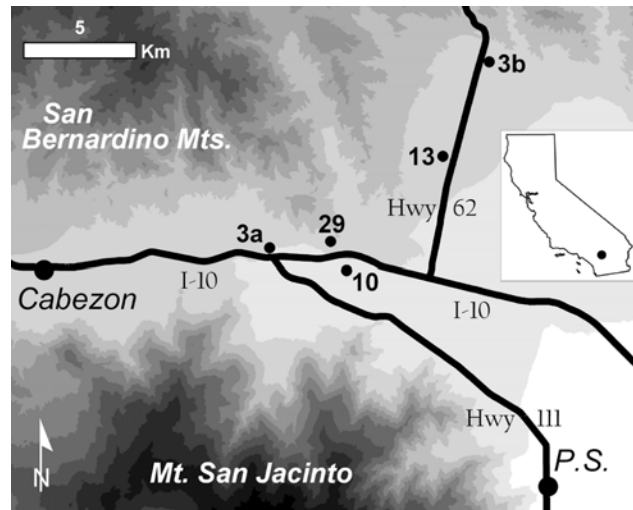


Figure 1. Study area on the western edge of Coachella Valley, Riverside County, California, showing study sites (numbers w/dots) that are named after their respective time since fire, as of spring 2008. P.S. represents the city of Palm Springs, I-10 is Interstate 10. Dark colors indicate increasing elevation.

Site Determination

In the spring of 2006, potential study sites were selected based on stereoscope validation of fire perimeters from a series of aerial photographs of the study landscape, spanning from 1949 to 2005. Aerial photos were obtained from Riverside County Flood Control and Water Conservation District, Coachella Valley Water District, and UC Riverside Science Library. The year when examined aerial photos were taken include the following: 1949, 1957, 1974, 1980, 1984, 1985, 1986, 1987, 1989, 1990, 1995, 1996, 1998, 2000, and 2005. Dates of the fires at each site were first determined from the aerial photos, but historic Los Angeles Times articles via ProQuest® (<http://www.proquest.com>), verbal communication with Richard Minnich (University of California, Riverside), and personal observations for all fires that occurred in 2005 were also used to date the year of fire. At two of the study sites utilized, the year of fire was only narrowed down to a 2 year period. Since fires in desert vegetation are more common following winter seasons with above average rainfall (Brooks and Matchett 2006), the wetter of the two possible burn years is reported in this experiment as the assumed burn year.

Respective unburned reference stands for all of the burned stands were also identified from aerial photographs in the spring of 2006. All paired unburned reference sites existed in similar areas of

shrub cover to pre-fire conditions based on aerial photographs. They were also located opposite fuel breaks (bulldozer lines, dirt roads, or paved roads) to minimize fuel differences at the time of each fire. Over twenty unique sites that had burned were identified in the study area from aerial photography, but after ground-truthing each site in July and August of 2006 only five were selected for this study. Sites dismissed from the study were done so mostly because of a lack of suitable unburned reference vegetation (in other words, unburned vegetation adjacent to burned vegetation did not appear to be separated by a fuel break so differences in fuel between the two areas may have existed at the time of fire). Other reasons for dismissal were because of recent grazing history (determined in the field), irregular soil type (based on NRSC soils maps or percent sand, silt, and clay analyses), or some sites were removed to minimize climatic variation (for example, sites adjacent to Cabazon or to Palm Springs).

The six sites selected for this study ranged in year burned from 1979 to 2005 with a time since fire (tsf) of 3, 3, 10, 13, and 29 years. The names of the sites correspond directly to tsf. The two sites that had burned in 2005 are called '3a' and '3b.'

Soil Sampling

In August and September of 2006, 6 sampling units were implemented in a stratified random design in both the unburned and paired burned reference vegetation at all six sites. Sampling units consisted of one modified – National Weed Management Association (mod-NAWMA) circular plot (Stohlgren and others 2003). Slope and aspect were measured from the center of each plot using a compass and clinometer. Soil was also collected for chemical and physical analyses. For nutrient analyses, four soil samples per mod-NAWMA plot were taken to 5 cm depth with a 2.5 cm diameter corer and pooled into one composite sample per plot. The four samples were taken at the center and at three edge locations (7.32 m from plot center), at 30, 150, and 270 degrees from plot center. One core with a 5 cm diameter was taken at the center of the plot for bulk density, coarse fraction (>2 mm), and soil texture measures. All soil sampled was taken at a 5 cm depth. Soil pH from the four pooled soil samples taken with the 2.5 cm diameter core was measured using a Fisher Scientific® Model 50 pH meter. The same soil samples were then analyzed for carbon (C)

nitrogen (N), NH_4^+ , and NO_3^- by the University of California, Davis, Analytical Laboratory (<http://groups.ucanr.org/danranlab>) in addition to percent sand, silt, and clay from the 5 cm diameter core.

Vegetation Sampling

In the winter wet-season of 2006-07, insufficient rainfall prevented the germination of annual plants at the study sites and no vegetation measurements were taken. In the wet season of 2007-08, precipitation was about average and vegetation was sampled throughout March 2008 during peak flowering in each established mod-NAWMA plot. Percent cover by species and species richness were measured in three 1 m² (1 x 1 m) quadrats per plot, located 4.57 m from plot center at 30, 150, and 270 degrees. Species richness was measured within each of the three 1 m² quadrats per plot and also within each plot (to a 7.32 m radius from plot center). All species nomenclature follows Hickman (1996).

Data Analyses

Vegetation cover at each of the five study sites was categorized into invasive grass, invasive forb, total invasive (grass + forb), native annual (grass + forb), herbaceous perennial, and native shrub cover at the 1 m² scale (in quadrats). For all analyses, shrub data included species in the Cactaceae. Species richness of invasive annuals, native annuals, and shrubs was also calculated at both the quadrat and mod-NAWMA plot scales. These parameters were used to compare the unburned stands of the six study sites with their respective, paired burned stands.

One-way ANOVA was used to compare the soil and vegetative variables between paired unburned and burned stands. When comparing shrub cover, Kruskal-Wallis tests were used instead of ANOVA, since these data were not normally distributed even when transformed. Two-way ANOVA was used to evaluate time since fire related differences in relative exotic grass, exotic forb, and native annual plant cover between paired burned and unburned stands. Lastly, a Detrended Correspondence Analysis (DCA) was performed for the two stands that had most recently burned (3a and 3b). Ordinations representing the other three sites were performed but are not reported because of weak (site 13) or no difference (sites 10 and 28) between paired stands. For each

DCA, only species that were recorded from three or more of the six plots from either paired stand (burned or unburned) were included. In other words, any species that was present in less than three plots per stand was removed prior to conducting the analyses. The DCA utilized mean species coverage values taken at the 1 m² scale and were used to compare plots based on their floristic composition as well as to determine which species were more associated with burned vegetation versus unburned (Vamstad and Rotenberry 2010). Analyses were performed using JMP 7.0.2 (SAS Institute Inc.) R v.11 (<http://www.r-project.org/>), and PC-ORD (McCune and Medford, 1999).

RESULTS

Vegetation of the Study Area

The vegetation of the study area was dominated by *Larrea tridentata*. *Ambrosia dumosa* was usually sub-dominant. Various shrubs and herbaceous perennials were documented in addition to seven exotic annuals and 46 native annual species (appendix 1). Exotic annual plant cover at all study sites was high (figure 2), which can be attributed to the invasive forbs *Erodium cicutarium* and *Brassica tournefortii*, and invasive annual grasses in the *Schismus* spp. complex (almost entirely represented by *S. barbatus*). *Bromus madritensis* ssp. *rubens* was common throughout the study area but was not one of the dominant annual plants, unlike prior years (Minnich 2008).

Impact of Fire on Soils

Most soil parameters did not differ between burned and unburned reference stands (table 1). Soil pH was greater in burned than unburned stands at the two most recently burned sites (3a and 3b). Other soil parameters, including total N, total C, and extractable nitrogen (NH₄⁺ and NO₃⁻), did not show consistent patterns between paired burned and unburned stands. Percent cover of bare ground in burned stands appeared to increase relative to paired unburned stands with time since fire. Litter cover had an opposite, albeit weak trend, where greater litter cover was found in unburned stands compared to burned stands at more recently burned sites, but differences in litter did not occur between paired stands at older burned sites (table 1).

Impact of Fire on Annual Plants

When analyzing absolute cover with One-way ANOVA, exotic grass cover was greater in burned than unburned vegetation at sites that had experienced fire three years ago, but not in older burned stands (figure 2). Exotic forb cover was only greater in burned than unburned vegetation at site 3b. Fire reduced native annual cover at the 1 m² scale at site 3a and 3b (figure 2).

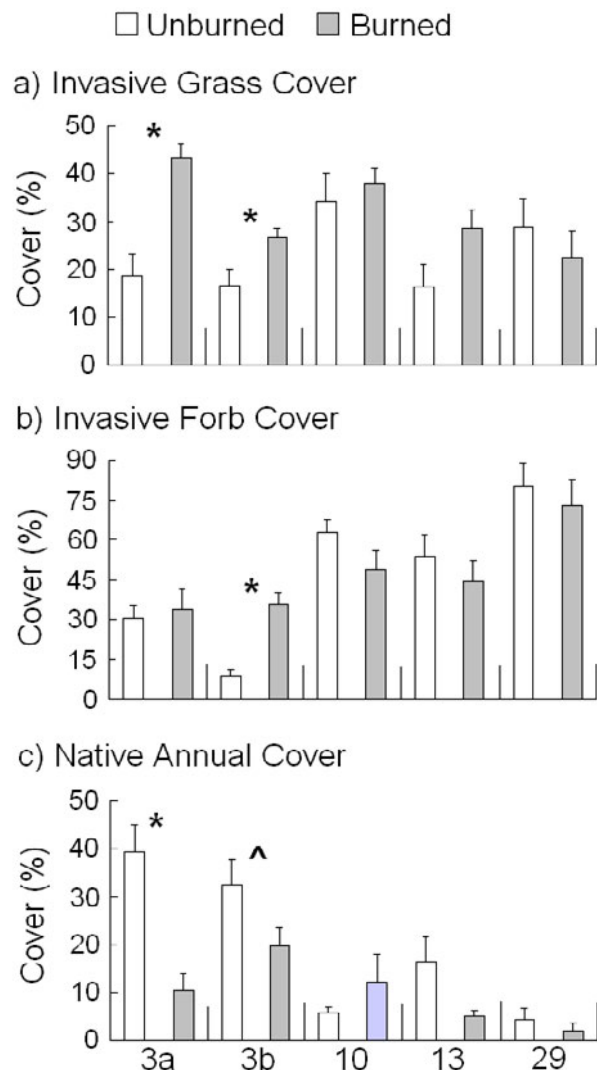


Figure 2. Mean invasive grass (a), invasive forb (b), and native cover (c) of annual plants. Statistical differences between paired stands per site are indicated by * ($P < 0.05$) or ^ ($P < 0.08$). Numbers below x-axis refer to site names, which also indicate years since burn.

When examining relative cover, recently burned stands had higher relative exotic grass cover but this decreased as time since fire increased, while exotic annual forbs became relatively more abundant as

time since fire increased. Relative cover of native annual plants decreased as time since fire increased (figure 3). Two-way ANOVA showed a significant effect of time since fire on relative exotic forb and native annual plant cover ($F = 17.8$, $p < 0.0001$; $F = 22.6$, $p < 0.0001$, respectively), a significant effect of fire on relative exotic grass and native annual plant cover ($F = 16.4$, $p = 0.0001$; $F = 23.4$, $p < 0.0001$, respectively) and a significant interaction between time since fire and whether the vegetation was burned or unburned for relative exotic grass, exotic forb, and native annual plant cover ($F = 4.8$, $p < 0.0048$; $F = 4$, $p < 0.0127$; $F = 7.1$, $p = 0.0004$, respectively).

Native annual plant richness at the 1 m² scale was also reduced by fire at sites 3b and 13 (figure 4). At the mod-NAWMA plot scale (168.3 m²), native annual richness was decreased by fire at sites 3a and 13 (figure 4). Site 3b, with the greatest species richness in unburned plots, surprisingly did not exhibit decreased native annual species richness in the burned stand at the scale of a mod-NAWMA plot. However, many of the species found in plots of the burned stand were only represented by one to a few individuals (Steers, personal observation during field sampling of study site, March 2008).

Ordinations produced with Detrended Correspondence Analysis (DCA) from the two recently burned sites showed a marked effect of fire on annual plant composition. For site 3a, the variance in the species data was 0.6434 and the eigenvalue for axis 1 was 0.35. Eigenvalues for the remaining two axes were less than 0.1 and 0.01, respectively. Thus, axis 1 explained most of the variation among plots. Unburned plots were positively associated with axis 1 while burned plots were negatively associated (figure 5). Thirteen annual plant species were included in the ordination (appendix 1). Species with the strongest positive association with axis 1 (indicative of unburned conditions) were *Bromus madritensis* ssp. *rubens* (axis score = 323), *Pholistoma membranaceum* (275), *Chaenactis fremontii* (194), and *Stephanomeria exigua* (176). Species most negatively associated with axis 1 (burned conditions) were *Erodium cicutarium* (-7 axis score), *Schismus* spp. (-4), *Crassula connata* (24), and *Filago californica* (88). The remaining species had scores ranging from 100 to 151.

Table 1. Mean soil and perennial plant parameters in unburned (UB) and paired burned (B) stands (numbers are site names and refer to year since burn; **3a**, and **3b** are two separate 3 year old sites). For each site, bold values indicate the stand with a significantly greater value among paired stands ($P = 0.05$). Aspect and slope were not statistically analyzed.

	3a		3b		10		13		29	
Soil and Vegetation Variables	UB	B	UB	B	UB	B	UB	B	UB	B
Aspect (deg.)	96	92	103	78	163	175	74	95	198	166
Slope (deg.)	1	1	2	1	2	2	1	1	4	4
Total N (%)	0.10	0.10	0.08	0.05	0.09	0.08	0.12	0.10	0.12	0.15
Total C (%)	0.84	0.89	0.76	0.56	0.99	0.75	1.07	0.83	1.33	1.62
NH ₄ ⁺ (ppm)	11.9	13.4	8.3	7.7	13.0	14.1	15.6	18.0	16.0	10.6
NO ₃ ⁻ (ppm)	28.1	18.0	7.6	7.6	19.1	14.5	13.9	9.9	16.1	17.2
Sand (%)	86	87	87	86	83	86	77	80	80	86
Silt (%)	12	11	12	12	14	11	19	16	15	11
Clay (%)	2	2	1	2	3	2	4	4	5	4
pH	7.7	7.8	7.3	7.6	7.2	7.4	7.4	7.5	7.0	7.0
Bulk Density (g · cm ⁻³)	1.36	1.61	1.6	1.57	1.43	1.58	1.14	1.24	1.36	1.25
Coarse Fraction (g · cm ⁻³)	0.14	0.16	0.25	0.11	0.26	0.23	0.4	0.46	0.48	0.48
Bare Ground Cover (%)	17	26	41	33	10	10	13	22	5	12
Rock Cover (%)	1	3	4	4	2	1	6	2	3	8
Litter Cover (%)	8	<1	3	2	1	1	3	1	2	3
Live Shrub Cover (%)	12	<1	8	<1	3	<1	7	1	7	13
<i>Encelia farinosa</i> Cover (%)	<1	<1	<1	<1			<1	1	<1	13
Shrub Richness (1 m ²)	0.4		0.6	0.1	0.2		0.4	0.4	0.2	0.5
Shrub Richness (168.3 m ²)	2.3	0.5	5.5	2	2.7	0.7	4.3	2.3	6.3	1.8

At site 3b, the variance in the species data was 1.1029 and the eigenvalue for axis 1 was 0.4116. Eigenvalues for the remaining two axes were 0.1099 and 0.0307, respectively. Again, axis 1 explained most of the variation among plots. Unburned plots were positively associated with axis 1 while burned plots were negatively associated (figure 5). Twenty four annual plant species were included in the ordination. Species with the strongest positive association with axis 1 (indicative of unburned conditions) were *Cryptantha barbiger* var. *fergusoniae* (323 axis score), *Chorizanthe brevicornu* (305), *Pholistoma membranaceum* (302), *Vulpia octoflora* (291), and *Bromus madritensis* ssp. *rubens* (248). Species most negatively associated with axis 1 (burned conditions) were *Plantago ovata* (-130), *Erodium cicutarium* (-83), *Pectocarya heterocarpa* (0), *Lepidium lasiocarpum* (41), *Malacothrix glabrata* (48), and *Schismus* spp. (71). All other species had scores ranging from 105 to 239, which are indicative of unburned conditions (figure 5).

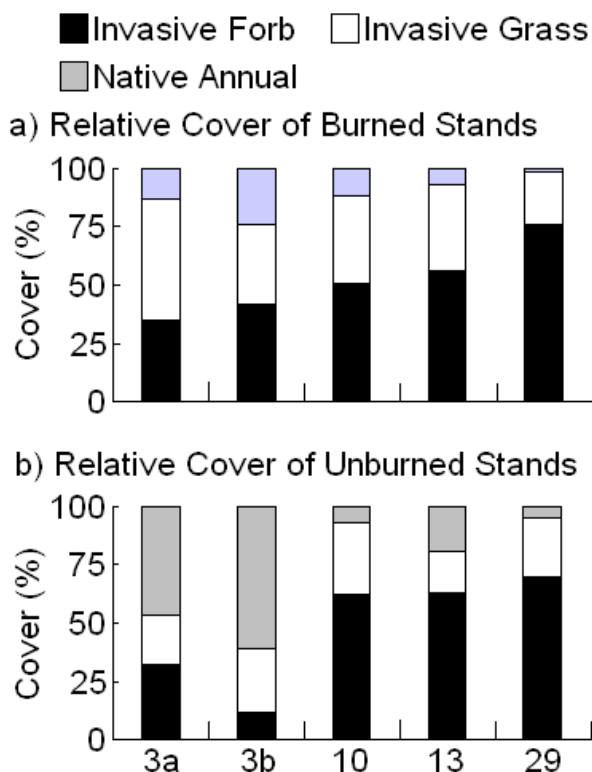


Figure 3. Relative cover of invasive annual forbs (black), invasive annual grasses (white), and native annual plants (grey) in paired unburned and burned stands. Numbers below x-axis refer to site names, which also indicate years since burn.

Impact of Fire on Perennial Plants

Live shrub cover was decreased by fires that had occurred three years prior to sampling (table 1). However, shrub cover did not show differences between burned and unburned vegetation by 10 years or more after fire. The increase in shrub cover in the two oldest burned stands (13 and 29) was due to recruitment by *Encelia farinosa* (table 1). Shrub cover was very low in both unburned and burned stands at site 10 (table 1); thus, no difference was detected at the 1 m² scale. Shrub richness was also compared between paired burned and unburned stands. At the 1 m² scale, only sites 3a and 3b experienced decreased shrub richness in the burned stands (table 1). However, at the larger, mod-NAWMA plot scale, shrub richness was reduced in burned stands compared to unburned reference stands at all five study sites (table 1).

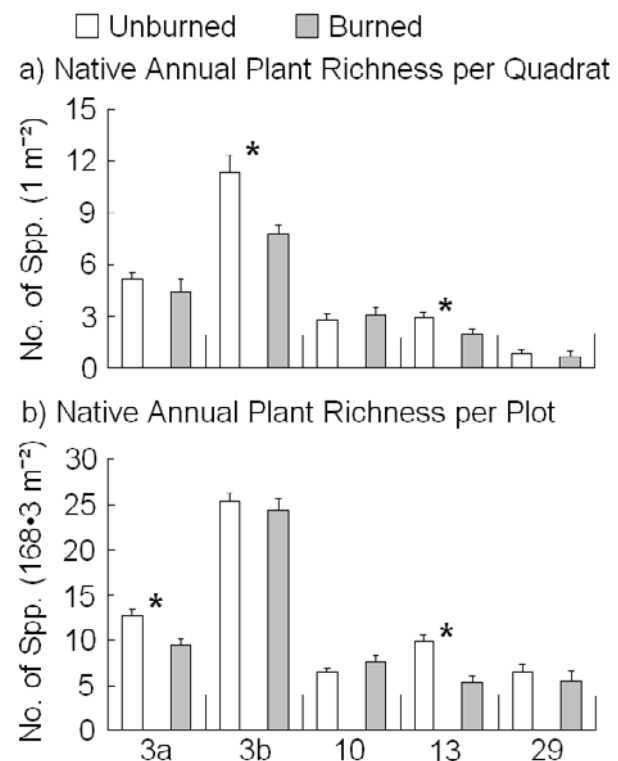


Figure 4. Mean species richness of native annual plants at each of the six study sites, in unburned and burned stands of CBS at the 1 m² quadrat (a) and mod-NAWMA plot (b) scales. * indicate significant differences between paired stands (P = 0.05).

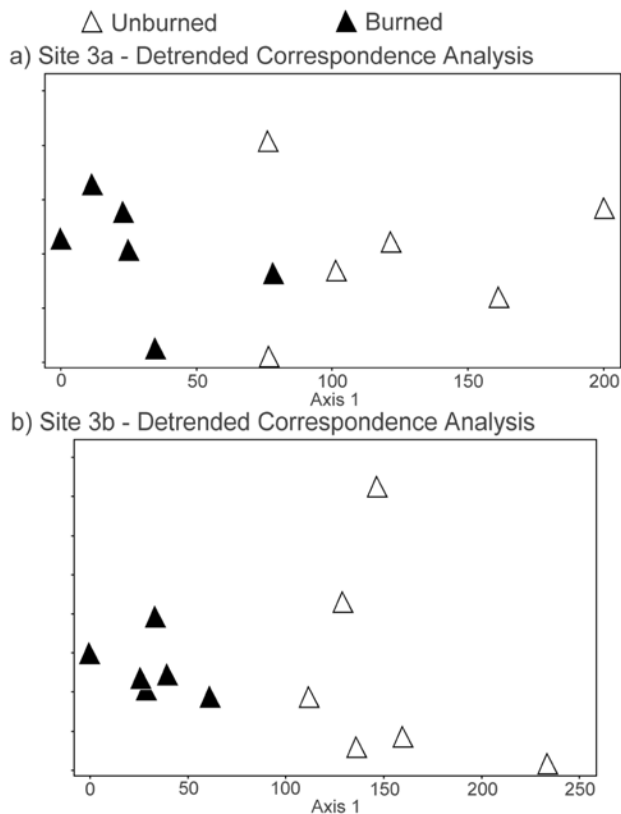


Figure 5. Ordinations of mean annual plant cover in burned and paired unburned plots from sites 3a and 3b using detrended correspondence analysis.

DISCUSSION

Impact of Fire on Soils

The lack of elevated NH_4^+ and NO_3^- in the two most recently burned stands was surprising, but may be explained because post-fire increases in these sources of N could be ephemeral and sampling soils three years after the burn may have missed any increase had it existed (Raison 1979). Soil pH increased after fire, which is consistent with other studies from different vegetation types (Raison 1979). In general, it appears that fires do not result in long-lasting effects on the soil parameters measured. However, the high cover of bare ground in the oldest burned stands may be indicative of higher soil erosion rates due to a desertification-like process (*sensu* Belnap 1995) that results when a site burns. Fires greatly reduced long-lived perennials, like *Larrea tridentata*, *Ambrosia dumosa*, and *Krameria greyi*, which are important for accumulating windblown soil and organic matter. Because the study area is affected by high winds (Rao and others 2011) and because *Encelia farinosa*, which became dominant, is not effective at accumulating organic matter in its

understory because of its architecture (Muller 1953), increased bare ground cover may be a result.

Impact of Fire on Vegetation

Fire increased exotic annual grass cover within the first three years after a fire (sites 3a and 3b), due almost entirely to *Schismus* spp, which has also been documented in other cases (Cave and Patten 1984; Minnich and Dezzani 1998; Brooks 2002). Fire also significantly increased exotic forb cover at site 3b, due to a non-significant increase in *Brassica tournefortii* and a significant increase in *Erodium cicutarium* (Steers 2008). Again, post-fire increases in *E. cicutarium* have been documented in other cases (Brown and Minnich 1986; Minnich and Dezzani 1998; Brooks and Matchett 2003). No other studies to our knowledge have reported a fire response by *B. tournefortii*; however, it appears this species is capable of responding positively. *Bromus madritensis* ssp. *rubens* typically decreases immediately following fire (Brooks 2002; Abella and others 2009) but no difference was detected in three year old burned stands in this study. This lack of response may be because this species was too infrequent and scarce for statistical analyses. Recent droughts have decreased *B. madritensis* across the landscape of the study area (Minnich 2008). At all other sites with 10 year or older burned stands, invasive annual vegetation did not differ between paired burned and unburned stands. The lack of difference between older paired stands was primarily because invasive plant abundance was very high in both paired stands. Had these older burned stands been sampled when they first burned, it is possible that significant differences could have been apparent.

Unfortunately, as invasive species become more abundant in new portions of the desert it is highly likely they will reduce native annual components of the vegetation. However, the ability of invasives to dramatically decrease native annual components of the vegetation in unburned conditions may be limited to regions of the desert where other factors positively associated with invasive annual plant abundance are elevated, such as nitrogen deposition, precipitation, and wind (Brooks and Esque 2002; Brooks and Matchett 2006; Brooks and Berry 2006; Rao and others 2010; Rao and others 2011). Other regions of the desert that are currently less impacted by these environmental factors may not suffer from invasives

and fire to the same extent as our study area, which has high wind and N deposition that disperse invasives and increase their productivity, coupled with precipitation amounts to produce sufficient fuel in many years (Rao and others 2010; Rao and others 2011) and elevated ignition sources (Brooks and Esque 2002).

Unfortunately, sites with the highest native annual plant abundance and richness experienced some of the largest relative increases in invasive plants once burned, and some of the greatest losses in native annual plant abundance and richness. Decreased annual species richness has been noted previously (Brooks 2002), but where *Schismus* spp. and *Erodium cicutarium* are less abundant or absent, fires can actually increase annual species richness immediately after fire (Abella and others 2009; Steers, unpublished data). Impacts to native annuals from fire, which have been observed under shrubs, have been attributed to lethal temperatures (Brooks 2002), but post-fire decreases in abundance and species richness of native annuals can also result from invasive plant competition (Brooks 2002; Steers and Allen 2010; in press).

Species responses to fire in the recently burned stands (3a and 3b) can be interpreted from DCA ordinations (Vamstad and Rotenberry 2010). In general, annual plant species that were not impacted or responded positively to fire were those species that seemed to be associated with inter-shrub spaces while species associated with the areas under shrubs were typically more impacted (R. Steers, personal observation during field sampling of study sites, March through April 2008). For example, *Bromus madritensis* ssp. *rubens*, *Phacelia distans*, and *Pholistoma membranaceum* were indicative of shrub understories and were associated with unburned plots. *Erodium cicutarium*, *Plantago ovata*, *Loeflingia squarrosa*, *Pectocarya* spp. and *Schismus* spp. were typical of intershrub spaces and were more closely associated with burned plots in the DCA. Post-fire increases in *Plantago ovata* have been documented previously (Brown and Minnich 1986; Cave and Patten 1984; Brooks 2002) but responses by the other annual species found in this study, besides exotic annuals, are largely unreported in the literature. Based on this study, it appears that once exotic annuals become abundant, native annuals decline, with or without fire, although fire can amplify this outcome. Lastly the impact of fire on perennial

components of the vegetation was severe and long-lasting, similar to findings from other studies (Abella 2009).

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Appendix 1. Species frequency during March 2008 sampling of creosote bush scrub vegetation of the study sites. Frequency of perennial and annual species found in the six mod-NAWMA plots per stand per site, with values varying from 0 to 6. Numbers refer to site names, which also indicate years since burn, UB = unburned stand, B = paired burned stand.

		3a		3b		10		13		29	
Family	Species	UB	B	UB	B	UB	B	UB	B	UB	B
HERBACEOUS PERENNIALS											
Asteraceae	<i>Stephanomeria pauciflora</i>				1		2	1	4		1
Cucurbitaceae	<i>Cucurbita palmata</i>				1						
Euphorbiaceae	<i>Chamaesyce polycarpa</i>				1					1	4
	<i>Ditaxis neomexicana</i>									1	1
	<i>Stillingia linearifolia</i>								1		
Liliaceae	<i>Dichelostemma capitatum</i>			1	2					3	
Nyctaginaceae	<i>Mirabilis bigelovii</i>			1	3		1	5	3	3	6
Poaceae	<i>Pleuraphis rigida</i>									1	
SHRUBS AND CACTI											
Asclepiadaceae	<i>Asclepias subulata</i>							1			
	<i>Acamptopappus</i>										
Asteraceae	<i>sphaerocephalus</i>							1			
	<i>Ambrosia dumosa</i>	6		6	6	3		5	5	6	1
	<i>Bebbia juncea</i>								2		
	<i>Encelia farinosa</i>	1	1	4	4			6	5	5	6
	<i>Hymenoclea salsola</i>			1					1		1
Cactaceae	<i>Echinocereus engelmannii</i>	1								1	
	<i>Ferocactus cylindraceus</i>							1		6	2
	<i>Mamillaria tetrancistra</i>									1	
	<i>Opuntia basilaris</i>									1	
	<i>O. bigelovii</i>									3	1
	<i>O. echinocarpa</i>			2						2	
Ephedraceae	<i>Ephedra californica</i>			5		1		1		1	
Fabaceae	<i>Psoralethamnus arborescens</i>			3	1						
Krameriaceae	<i>Krameria grayi</i>			4		6		4		6	
Liliaceae	<i>Yucca schidigera</i>			2							
Polygonaceae	<i>Eriogonum fasciculatum</i>							1			
Rutaceae	<i>Thamnosma montana</i>							1			
Zygophyllaceae	<i>Larrea tridentata</i>	6	2	6	1	6	4	6		6	
INVASIVE FORBS											
Asteraceae	<i>Sonchus oleraceus</i>									1	
Brassicaceae	<i>Brassica tournefortii</i>	6	6	6	6	6	6	6	6	6	6
Geraniaceae	<i>Erodium cicutarium</i>	6	6	6	6	6	6	6	6	6	6
INVASIVE GRASSES											
Poaceae	<i>Bromus diandrus</i>					1					
	<i>B. madritensis</i>	4		6	5	1		6	2	4	1
	<i>Hordeum murinum</i>	1	1								
	<i>Schismus spp.</i>	6	6	6	6	6	6	6	6	6	6
NATIVE ANNUALS											
Asteraceae	<i>Chaenactis fremontii</i>	6	6	6	6	1	1	3	1	3	4
	<i>Eriophyllum wallacei</i>		1	1	1						
	<i>Filago californica</i>	6	6	1		6	6			6	4
	<i>F. depressa</i>			6	6						

Appendix 1 (cont). Species frequency during March 2008 sampling of creosote bush scrub vegetation of the study sites.

Asteraceae	<i>Lasthenia californica</i>	1						1	1		
	<i>L. coronaria</i>			1	2						
	<i>Malacothrix glabrata</i>	6	6	6	6	6	5	4	2	2	3
	<i>Rafinesquia neomexicana</i>	2		6	5			3			
	<i>Stephanomeria exigua</i>	5	3		1					1	2
	<i>Stylocline gnaphaloides</i>	1		4	4						
Boraginaceae	<i>Cryptantha angustifolia</i>				3						
	<i>C. barbiger</i> var. <i>fergusoniae</i>	2		6	6			6	4	1	6
	<i>C. circumscissa</i>			3	3						
	<i>C. decipiens</i>			1							
	<i>C. maritima</i>			5				1			
	<i>C. micrantha</i>	1		3	5						
	<i>C. nevadensis</i>			1							
	<i>Pectocarya heterocarpa</i>	3	4	6	6	2	6	4	6	1	
	<i>P. linearis</i>	6	5	6	2	6	6	6	6	4	3
	<i>P. platycarpa</i>			4							
	<i>P. recurvata</i>	3	1	6	6			6	6	3	1
Brassicaceae	<i>Lepidium lasiocarpum</i>			6	4			1	1	1	
	<i>Tropidocarpum gracile</i>							1		4	1
Campanulaceae	<i>Nemacladus longiflorus</i>				1						
Caryophyllaceae	<i>Loeflingia squarrosa</i>	2	5	6	5		4				
Crassulaceae	<i>Crassula connata</i>	6	6	6	6	6	6	6	4	6	6
Fabaceae	<i>Lotus strigosus</i>	6	4	4	5	6	6			2	1
Hydrophyllaceae	<i>Emmenanthe penduliflora</i>				3						
	<i>P. distans</i>	1		6	6			3			
	<i>Pholistoma membranaceum</i>	6		6	3			6		5	
Lamiaceae	<i>Salvia columbariae</i>			1				1			
Loasaceae	<i>Mentzelia</i> sp.			2	1			1			
Onagraceae	<i>Camissonia californica</i>			5	6						1
	<i>C. pallida</i>	6	6	6	6	2	3	1			1
Papaveraceae	<i>Eschscholzia minutiflora</i>				2						
Plantaginaceae	<i>Plantago ovata</i>	1			6						
Poaceae	<i>V. octoflora</i>	6	1	6	5	4	3	1			
Polemoniaceae	<i>Eriastrum diffusum</i>			3	3						
	<i>Eriastrum</i> sp.			3	3						
	<i>G. maculata</i>			2							
	<i>Linanthus bigelovii</i>			1							
	<i>Loeseliastrum schottii</i>			3	2						
Polygonaceae	<i>Chorizanthe brevicornu</i>	1	2	6	6			4	1		
	<i>C. watsonii</i>				2						
	<i>Pterostegia drymarioides</i>			2	3						
Portulacaceae	<i>Calyptidium monandrum</i>			6	6						

Bottom-up Effects of Substrate on Two Adjacent Shrub Communities and the Distribution of a Rare and Endangered Plant Species, *Astragalus jaegerianus* Munz.

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ABSTRACT

Edaphic habitats are botanically interesting because of differences in vegetation with neighboring sites and because they tend to harbor rare species. In the central Mojave Desert, there are granite colluvial substrates where creosote bush, the dominant shrub in the area, is sparser and generally smaller than in the neighboring creosote bush communities. It is on these sites that the Lane Mountain milkvetch, a rare and federally endangered species, is restricted. The milkvetch is a nitrogen-fixer and grows under and within the canopy of host shrubs. Our previous studies have demonstrated that the milkvetch has no preference for species of host shrub, except Larrea tridentata, which appears to be an unsuitable host plant for the milkvetch. In this study, we surveyed three transects within milkvetch habitats and three transects in adjacent creosote bush habitats in the year 2000 and again in 2010, a period coincident with long-term drought conditions in the Mojave Desert. Our results show that adjacent milkvetch and creosote bush shrub communities differ significantly in shrub height, shrub volume, and shrub density in the year 2000: the shrubs in milkvetch communities were more numerous but smaller compared to adjacent creosote bush scrub. Species richness also differed between communities in the year 2000: milkvetch communities contained 19 different shrub species and creosote bush communities had only 9 species. Surveys in 2010 show that the drought had significant negative effects on both shrub communities. Total shrub mortality (166 shrubs) was high compared to shrub recruitment (16 shrubs), and the majority of mortality and recruitment occurred in milkvetch communities (131 deaths and 16 recruits). Shrub densities decreased significantly in milkvetch communities in 2010, but were still considerably higher than in creosote bush communities. These results suggest that the restricted distribution of the Lane Mountain milkvetch may be the result of higher shrub densities in milkvetch shrub communities; increased shrub densities increases the proximity of suitable host shrubs, which in turn increase the probability of successful seed dispersal and establishment.

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INTRODUCTION

Ecosystems often contain dominant plant communities composed of species that attain high densities within the prevailing edaphic and climatic conditions associated with that ecosystem. Within these dominant plant communities, abrupt changes in soil characteristics can create islands of distinctive vegetation in which the regionally dominant vegetation is excluded or modified (Kruckeberg and Rabinowitz 1985; Mason 1946). These anomalous habitats may be geographically isolated and of limited areal extent, and they are usually less productive than the surrounding, regionally dominant vegetation (Meyer 1986; Whittaker 1954). These anomalous edaphic communities are botanically interesting

because they demonstrate the bottom-up effect of soils on plant communities, and because they tend to harbor unusual species that may be rare and endemic, or represent major disjunct populations (Kruckeberg and Rabinowitz 1985; Gankin and Major 1964; Whittaker 1954).

The soil properties responsible for vegetation differences within edaphic communities may include the presence of elements toxic to the physiology of most plants (for example, magnesium in serpentine soils (Proctor 1970), or calcium and aluminum in limestone soils (Lee 1999)). Vegetation differences may be due to soil deficiency in iron or calcium (Brady and others 2005; Lee 1999), or deficiency in the essential nutrients nitrogen, potassium, and

phosphorous (Zohlen and Tyler 2004; Proctor and Woodell 1975). Vegetation differences in edaphic communities may also be due to deficiencies in soil moisture (Ware 1991; Baskin and others 1972). In some cases the soil properties responsible for vegetation differences in edaphic communities are complicated and not fully understood (for example, gypsum plants; Palacio and others 2007; Meyer 1986). Some edaphic communities may serve as refugia for plants that can tolerate toxic compounds (in other words, the refugia hypothesis, Gankin and Major 1964), while other edaphic communities may contain some feature essential for an edaphic species (in other words, the specialist hypothesis, Meyer 1986).



(a)



(b)

Figure 1. Adjacent plant communities at Gemini Conservation Area in January 2010, Fort Irwin, CA (elevation 1110 m). A. Creosote bush-dominated community with elements of Joshua Tree woodlands including *Yucca brevifolia* (tree in the left middle-distance). The abundant, large, dark shrubs are creosote bush (*Larrea tridentata*). B. Milkvetch granite barren, with its characteristic high density of small shrubs in which creosote bush is absent or reduced. The dark shrub in the right-foreground is a lone creosote bush.

Astragalus jaegerianus Munz (Fabaceae), the Lane Mountain milkvetch, is a narrowly endemic, herbaceous perennial restricted to rocky granite outcrops (barrens) in the central Mojave Desert. These granite barrens occur at an elevation between 900 and 1200 m, and are easily recognized by their reduced vegetation compared to the adjacent dominant vegetation of the area, creosote bush scrub, a shrub community dominated by *Larrea tridentata* (D.C.) Cov. (creosote bush) with common associates including *Ambrosia dumosa* (A. Gray) W. W. Payne (burro-weed), *Krameria erecta* Schult. (pima rhatany), *Ephedra nevadensis* S. Wats. (Nevada ephedra), and *Grayia spinosa* (Hook.) Moq. (spiny hopsage), as well as the *Yucca brevifolia* Engelm. (Joshua tree). *Larrea tridentata*, which is dominant in the adjacent vegetation (figure 1A), is conspicuously absent or much reduced within the shrub community on granite barrens (figure 1B). Similarly, *A. jaegerianus* does not occur outside of granite barrens in adjacent creosote bush scrub. The shrub community is important to *A. jaegerianus*, a climbing herbaceous perennial, because it uses shrubs as host plants (Gibson and others 1998, Huggins and others 2010). *Astragalus jaegerianus* completes its entire lifecycle within its host shrub; it germinates or resprouts under the shrub canopy in winter, then climbs through the interior of the shrub emerging onto the canopy where it flowers and sets fruits in late spring. *Astragalus jaegerianus* then goes dormant through the summer until it resprouts again with winter rains.

The central purpose of this study is to investigate the factors that act to restrict *A. jaegerianus* to granite barrens of the central Mojave Desert. To explore this phenomenon we (1) describe the edaphics and vegetation of shrub communities on granite barrens and adjacent creosote bush scrub, (2) analyze *A. jaegerianus*' host shrub preferences, and (3) propose a hypothesis explaining the restricted distribution of *Astragalus jaegerianus* on granite barrens as a function of shrub density and size. In addition, we (4) describe how recent drought conditions in the central Mojave have affected *A. jaegerianus* and the shrub in granite barrens and creosote bush scrub communities. Severe drought conditions in the Mojave began in 1999 and are predicted to continue for decades (Hereford and others 2006), or may

continue indefinitely under warmer temperature conditions projected by global climate change-type drought (Cook and others 2004; Breshears and others 2005; Seager and others 2007). These recent drought conditions have led to unusually high shrub mortality and canopy dieback in the Mojave Desert and other parts of the arid southwest US (Bowers 2005; Miriti and others 2007; Hamerlynnck and McAuliffe 2008; Hamerlynnck and Huxman 2009).

METHODS

Study Site

Astragalus jaegerianus exists in small fragmented populations within an area of less than 240 km². Roughly two-thirds of all known *A. jaegerianus* populations occur within the boundaries of the US Army's National Training Center at Fort Irwin, approximately 50 km NE of Barstow, California (Charis 2002). The *A. jaegerianus* populations described in this study occurred within Brinkman Wash and the Gemini Conservation area (previously Goldstone), one of four locations previously established as discrete areas of *A. jaegerianus* distribution (Charis 2002; Walker and Metcalf 2008). The soils at these sites are composed of shallow granitic colluvium on rocky, granitic outcrops, within the transition zone between Mojavean creosote bush scrub and Joshua tree woodland communities (sensu. Thorne 1982). Adjacent creosote bush communities occur on deeper (greater than 1 m) granite alluvium substrates.

Vegetation

In 1999, five 1-ha plots on granite barrens within the Brinkman Wash area were systematically searched for *A. jaegerianus*. The shrub communities within these plots were visually similar to other shrub communities supporting *A. jaegerianus* across its range. Shrubs harboring *A. jaegerianus* were marked and their UTM coordinates recorded using GPS. The species identity of these host shrubs was also noted to determine *A. jaegerianus* host preferences. Shrub density, cover, and volume within the five granite barren plots were sampled using four belt-transects per plot (Mueller-Dombois and Ellenberg 1974). For

each of these belt-transects in granite barrens, four additional belt-transects were sampled in near-by or adjacent creosote bush scrub, in order to compare both shrub communities. The belt-transects were 24 m long, and either 2 m wide for sampling small shrubs, or 4 m wide for sampling *Larrea tridentata* and *Yucca brevifolia*. The belt-transects were permanently marked using iron rebar stakes and UTM coordinates were recorded. Density was determined by tallying all shrubs in the belt-transect, and cover and volume was determined for all small shrubs in the first 12 m of the belt-transect and for all *L. tridentata* and *Y. brevifolia* for the total length of the belt-transect. Cover was calculated using the formula for an ellipse (ellipse area = $\pi d_1 d_2 / 4$ where d_1 = maximum shrub diameter and d_2 = diameter perpendicular to the maximum diameter), and volume for an ellipsoid (volume = area * h / 2, where h = height of shrub). In 2010, three of the five granite barren plots and their associated creosote bush scrub transects were re-sampled to determine the response of each shrub community to the drought conditions which began in 1999.

In addition to the five study sites described above, two more 1 ha study plots were established in 2003 in the Gemini Conservation Area approximately 6 km north of Brinkman Wash study plots. These Gemini study plots were intended for long-term monitoring of *A. jaegerianus* populations, and together with two of the Brinkman Wash study plots established in 1999 were surveyed annually starting in 2003. Like the Brinkman Wash study plots, each shrub within the 1-ha Gemini study plots was systematically search for *A. jaegerianus*, and shrubs harboring *A. jaegerianus* were marked, their UTM coordinates recorded, and the identity of host shrubs was noted. The Gemini study plots were not part of the Brinkman Wash shrub vegetation study, and so contained no belt transects.

Host Shrubs

Astragalus jaegerianus uses various shrub species as hosts (table 1). To determine whether these different values for host shrubs represent preferences or merely reflect the relative abundances of shrub species on granite barrens, a total of the observed species used by *A. jaegerianus* within all five 1 ha plots was compared to an expected value based on

shrub relative densities calculated from the belt transects. The expected values (expected value = relative density of each shrub species $\times$ number of shrubs with *A. jaegerianus* at the site sampled) were calculated for each shrub species in each of the five study sites, and the expected values for each species for each study site were summed. A goodness of fit analysis (Zar 1974) was performed to determine if observed values for a host were statistically different from expected values. Many of the expected values for the less common shrub species had to be combined together in "other species" to meet the recommendations of Cochran (1954): no expected frequency should be less than 1.0 and no more than 20 percent of the expected frequencies should be less than 5.0. Because the initial goodness of fit analysis had a significant chi-square value, subdivisions of the goodness of fit analysis (Zar 1974) were performed on subdivided data sets. *Larrea tridentata* had a very large partial chi-square value, so a goodness of fit was performed on the data set "*L. tridentata* versus all

other shrubs" and another analysis on the data set of shrubs excluding *L. tridentata*.

Edaphic Analysis

Soil pits were dug at milkvetch sites and at a neighboring creosote bush scrub site. Pits were dug to the bedrock at milkvetch granite barren sites (5 to 40 cm) and at the creosote bush scrub sites to ca. 80 cm deep. The soil depth to impervious layers (if any) was noted. Each soil sample was placed in a re-sealable plastic bag and brought back to UCLA where they were immediately air dried. Samples were then gently pulverized to break up aggregates and then sieved to remove particles > 2 mm. The samples were analyzed by the Agriculture and Natural Resources Analytical Laboratory at the University of California at Davis for particle size distribution (sand, silt, and clay), organic content, pH, salts (Ca, Mg, Na, electrical conductivity), nutrients [N (N_{TKN}), NH_4 -N, NO_3 -N] P, K], selenium, iron, and cation exchange capacity.

Table 1. The change in abundance of host shrubs with live *A. jaegerianus* at Brinkman Wash (1999-2010) and the Gemini Conservation Area (2003-2010). *Astragalus jaegerianus* may survive after its host shrub has died (dead shrub), and is rarely found growing without a host shrub (no host). *Astragalus jaegerianus* was not found growing within *Larrea tridentata*. Brinkman Wash was first surveyed in 1999, and the Gemini Conservation Area in 2003.

Brinkman Wash			Gemini Conservation Area		
Host	1999	2010	Host	2003	2010
<i>Eriogonum fasciculatum</i>	21	2	<i>Eriogonum fasciculatum</i>	20	1
<i>Thamnosma montana</i>	18	4	<i>Thamnosma montana</i>	6	1
dead shrub	16	1	<i>Ericameria cooperi</i>	6	0
<i>Ephedra nevadensis</i>	15	1	<i>Ambrosia dumosa</i>	3	3
<i>Salazaria mexicana</i>	8	1	<i>Ephedra nevadensis</i>	2	2
<i>Ericameria teretifolia</i>	7	2	<i>Ambrosia salsola</i>	2	0
<i>Ericameria cooperi</i>	6	0	<i>Encelia actonii</i>	2	0
<i>Ambrosia dumosa</i>	6	1	<i>Xylorhiza tortifolia</i>	1	0
<i>Xylorhiza tortifolia</i>	5	0	<i>Krameria erecta</i>	1	0
<i>Ambrosia salsola</i>	4	1	<i>Ericameria teretifolia</i>	0	0
no host	2	0	<i>Salazaria mexicana</i>	0	0
<i>Krameria erecta</i>	1	0	dead shrub	0	0
<i>Encelia actonii</i>	0	0	no host	0	0
<i>Larrea tridentata</i>	0	0	<i>Larrea tridentata</i>	0	0
Total	109	13	Total	43	7

RESULTS

Edaphics

The pooled results of the edaphic analysis are presented in table 2. Differences in the origin of soils within granite barrens and creosote bush-dominant communities result in soils of different depths. Soils within granite barrens are a product of granite decomposition within the granite barrens themselves (colluvial). Consequently, soils within granite barrens are shallow, with parent granite no deeper than 40 cm from the soil surface, but commonly shallower, with exposed granite visible on low alternating ridges within the granite barrens (figure 2). Soils within adjacent creosote bush scrub are composed of alluvium from the surrounding hills including the low ridges within the milkvetch granite barrens.

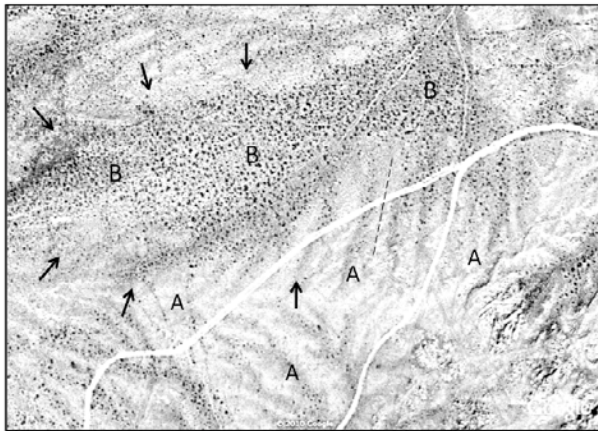


Figure 2. A Google Earth satellite image of typical milkvetch-granite barrens (A) with adjacent creosote bush scrub (B) at the Gemini Conservation Area (elev. 1110 m) within the National Training Center at Fort Irwin, CA. Creosote bushes are clearly visible as dark spots on deep alluvial soils that support creosote bush scrub (B). Visible within the milkvetch barrens (A) are the exposed granite ridges typical of milkvetch habitat (- - -) where creosote bush is absent or reduced in size and density. Soil depths in milkvetch granite barrens may reach 30 cm in drainages between ridges. Arrows indicate the direction of run-off from elevated milkvetch granite barrens into lower areas where alluvium accumulates producing deep soils (>1 m) that support creosote bush scrub.

Consequently, soils with creosote bush scrub are deep, typically greater than 1 m in depth. Some physical and chemical properties of the soil of milkvetch granite barrens and adjacent creosote bush scrub are presented in table 2. Soil textures within the creosote bush scrub and milkvetch granite barrens

are similar and typical of Mojave Desert soils composed primarily of sand (Stevenson and others 2009; Young and others 2009). Electrical conductivity, CaCO_3 , and pH are more or less similar between communities, and within the range of typical Mojave Desert soils but at the low end of the range (Graham and O'Geen 2010; Cox and others 1984, Romney and others 1973). Both nitrogen (total Kjeldahl N and total available N ($\text{NH}_4\text{-N}$ plus $\text{NO}_3\text{-N}$)), phosphorous and potassium are somewhat lower in granite barrens than in adjacent creosote bush scrub communities (table 2), but both communities have values higher than other Mojave Deserts creosote bush sites (Schlesinger and others 1996), and within the range of typical Mojave Desert soils (Schlesinger and others 1996; Romney and others 1973; Cox and others 1984; Rundel and Gibson 1996). Like NPK, elements such as calcium (Ca), sodium (Na), and selenium (Se) are somewhat higher in the alluvial creosote bush sites. These higher levels are likely to be the result of the accumulation and concentration of particles dissolved in run-off from the higher elevations surrounding creosote bush scrub sites. The slightly higher organic content in creosote bush scrub may account for slightly higher nitrogen in creosote bush scrub soils. In both communities, iron (Fe) occurs at levels higher than reported in other Mojave Desert soils (0 to 2.0 ppm reported in Romney and others 1973 for soils of the Nevada Test Site), and occurs at higher levels in milkvetch granite barrens (4.0 ppm Fe) than in creosote bush scrub (2.3 ppm, table 2), but both values are well below concentrations that are typically toxic to plants (> 365 ppm, Foy and others 1978).

Host Plants

The chi-square test for goodness of fit for all shrubs (table 3) demonstrates that the observed species spectrum of host shrub frequencies does not fit expected frequencies based on our belt transect data (chi-square = 20.68; critical value = 7.81). One species alone, *Larrea tridentata*, constitutes over 57 percent of the chi-square value, and is significantly under-represented as an *A. jaegerianus* host plant (chi-square = 11.98; critical value = 3.84). This result suggests that *Larrea* may be an unsuitable host shrub for *A. jaegerianus*. This unsuitability may be due to *Larrea*'s architecture, which is significantly taller than suitable host shrubs, and too tall and open to allow *A. jaegerianus* to climb through and reach its canopy.

Table 2. Physical and chemical properties of substrate at milkvetch granite barrens and creosote bush scrub sites at Brinkman Wash, 2001. Twenty-six soil samples were collected per site. Samples were collected under and between shrub canopies, at a depth of 5 to 40 cm. Values are means (SE).

Substrate Characteristics	Sites	
	Milkvetch Granite Barrens	Creosote Bush Scrub
General Properties		
	colluvium, decomposed granite	alluvium, some sites derived 1° from granite, other sites a mixture derived from granite and sedimentary rock
Physical Properties		
Depth	shallow - parent rock within 40 cm of surface	deep - hard pan deeper than 100 cm of surface
Texture		
Sand %	77 (0.9)	75 (1.3)
Slit %	15 (0.8)	18 (0.9)
Clay %	9 (0.4)	7 (0.6)
Chemical Properties		
Organic content (%)	0.5 (0.06)	0.8 (0.1)
pH	7.4 (0.1)	7.7 (0.1)
Electrical conductivity	0.35 (0.04)	0.86 (0.2)
Ca (meq/L)	2.5 (0.4)	6.9 (1.3)
CaCO ₃ (%)	0.58 (0.2)	0.54 (0.1)
Mg (meq/L)	0.68 (0.1)	0.93 (0.2)
Na (meq/L)	0.31 (0.03)	0.50 (0.07)
Se (ppb)	43.2 (5.2)	144.2 (19.3)
Fe (ppm)	4.0 (0.5)	2.3 (0.19)
Cation exchange capacity	17.0 (0.7)	17.9 (0.8)
N _(Total Kjeldahl N) (%)	0.038 (0.01)	0.060 (0.01)
K (ppm)	11.3 (3.3)	45.0 (10.8)
P (ppm)	9.8 (1.4)	16.2 (2.1)
NH ₄ -N (ppm)	2.2 (0.6)	1.8 (0.3)
NO ₃ -N (ppm)	4.4 (0.9)	15.8 (4.5)
N _(available) (ppm)	6.6 (1.5)	17.6 (4.8)

Excluding *Larrea*, a second, goodness-of-fit test with the remaining shrubs is not significantly different from expected (chi-square = 9.99; critical value = 11.07). Thus, *Eriogonum fasciculatum* Benth. var. *polifolium* (Benth.) Torr. & A. Gray (Mojave buckwheat), *Ericameria cooper* (A. Gray) H. M. Hall (Cooper's goldenbush), *Ephedra nevadensis*, *Ambrosia dumosa*, and *Krameria erecta* are used as host plants in what would be expected from their abundances on the sites. The category "other shrubs," which includes *Encelia actonii* Elmer (Acton's encelia), *Salazaria mexicana* Torr. (Mexican bladder sage), *Ericameria teretifolia* (Durand & Hilg.) Jeps. (green rabbitbrush), *Ambrosia salsola* (Torr. & A. Gray) Strother & B.G.

Baldwin (cheesebush), *Thamnosma montana* Torr. & Frem. (turpentine broom), and *Xylorhiza tortifolia* (Torr. & Gray) Greene (Mojave woody aster), were also used as hosts in the same frequency their abundance would suggests but their sample size it too small to reach any statistically valid result.

Vegetation

As expected from visual differences between communities, shrub characteristics on the granite barrens were very different from adjacent creosote bush scrub (figure 3): shrubs on granite barrens were smaller (0.062 m³ versus 0.86 m³), shorter (27 versus 67 cm), and produced less cover than shrubs in

creosote bush scrub (17 percent versus 25 percent), but were more numerous, resulting in a higher density of shrubs on granite barrens (181 versus 74 shrubs/100 m²). At our study sites, *Larrea* is a taller and larger shrub than *A. jaegerianus* host shrub species (figure 4), and its density is much higher in creosote bush scrub than in milkvetch-granite barrens (figure 5). Since *Larrea* is a large, unsuitable host for *A. jaegerianus* (table 3), its dominance in cover within creosote bush scrub preempts space used by potential host shrubs, further reducing the effective density of host shrubs in creosote bush scrub.

Twenty-five species of perennial shrubs occurred within the belt transects in granite barrens and adjacent creosote bush (figure 6). Transects in both communities supported similar species richness

values; 20 species in granite barrens and 19 species in creosote bush scrub. Of these 25 species, and ignoring the rarer species (< 0.01 percent absolute cover), five species occurred exclusively within granite barren transects, and two species occurred exclusively within creosote bush scrub transects. Three species were common in creosote bush scrub: *Larrea tridentata*, *Ambrosia dumosa*, and *Krameria erecta*, with *Larrea* dominating in absolute (percent) cover. Four species were co-dominants in granite barrens: *Larrea tridentata*, *Eriogonum fasciculatum*, *Ericameria cooperi* and *Ephedra nevadensis*. These four co-dominants, with the exception of *Larrea*, are common *A. jaegerianus* host shrubs, but represent a minute component of the adjacent creosote bush scrub communities.

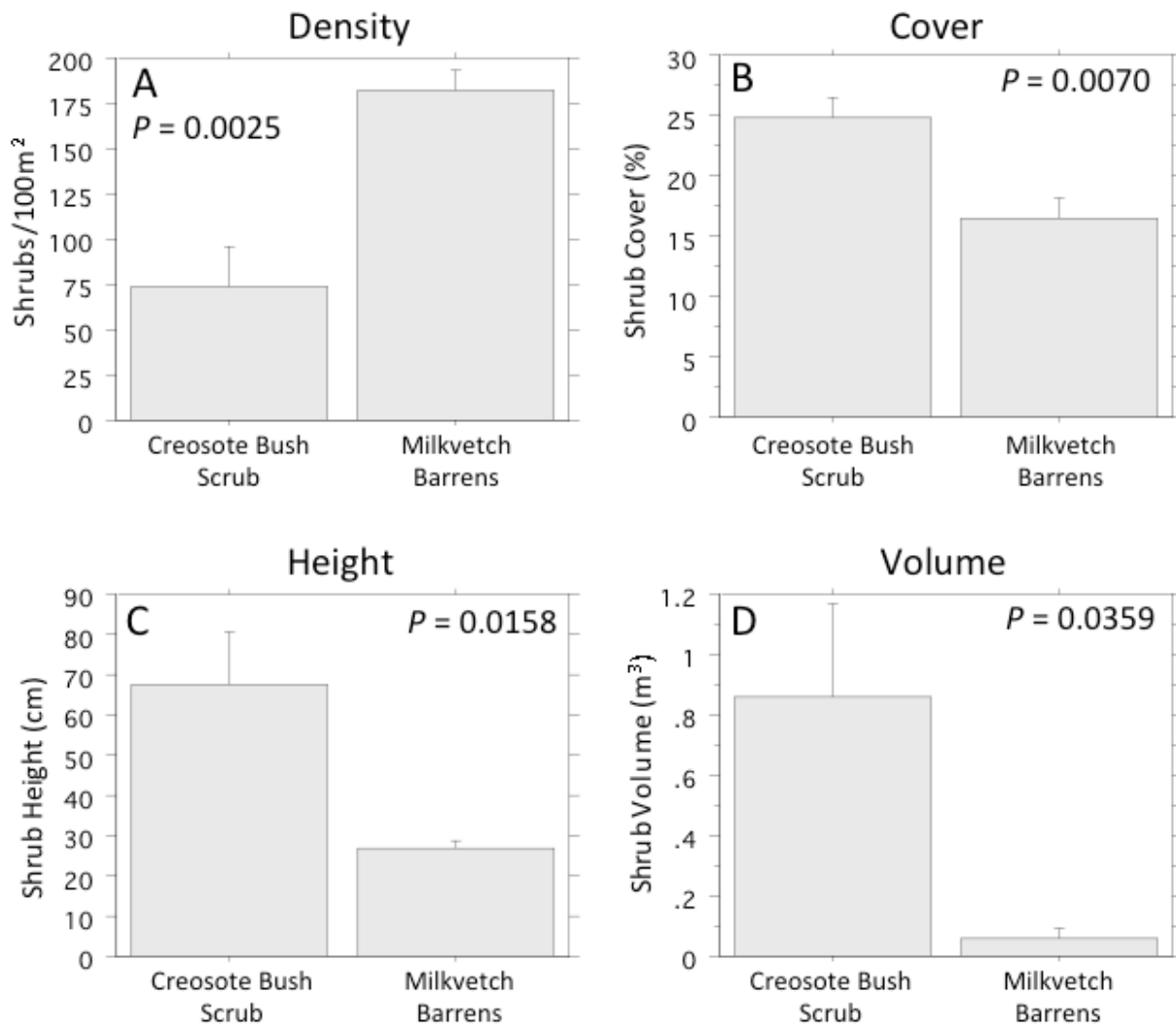


Figure 3. Differences in the distribution and size of shrubs in adjacent shrub communities. Shrubs are significantly larger in creosote bush scrub (C and D) and cover a greater area (B), but shrubs are significantly more numerous in milkvetch granite barrens (A). Bars are means (+ SE).

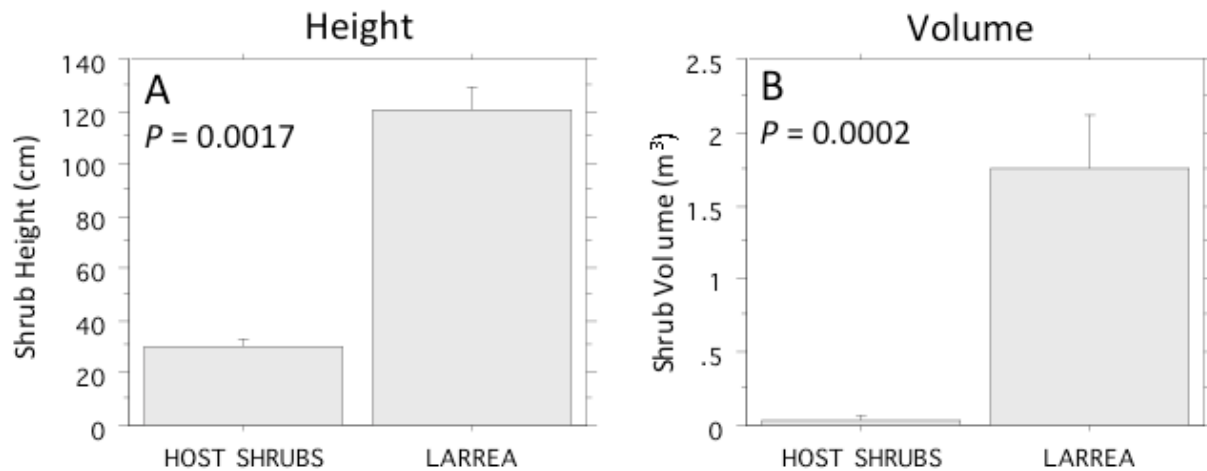


Figure 4. Differences in the size of typical *Astragalus jaegerianus* host shrub species and *Larrea tridentata* (creosote bush), an unsuitable host shrub for *A. jaegerianus*. *Larrea tridentata* shrubs are significantly taller (A) and larger (B) than typical *A. jaegerianus* host shrubs. Bars are means (+ SE).

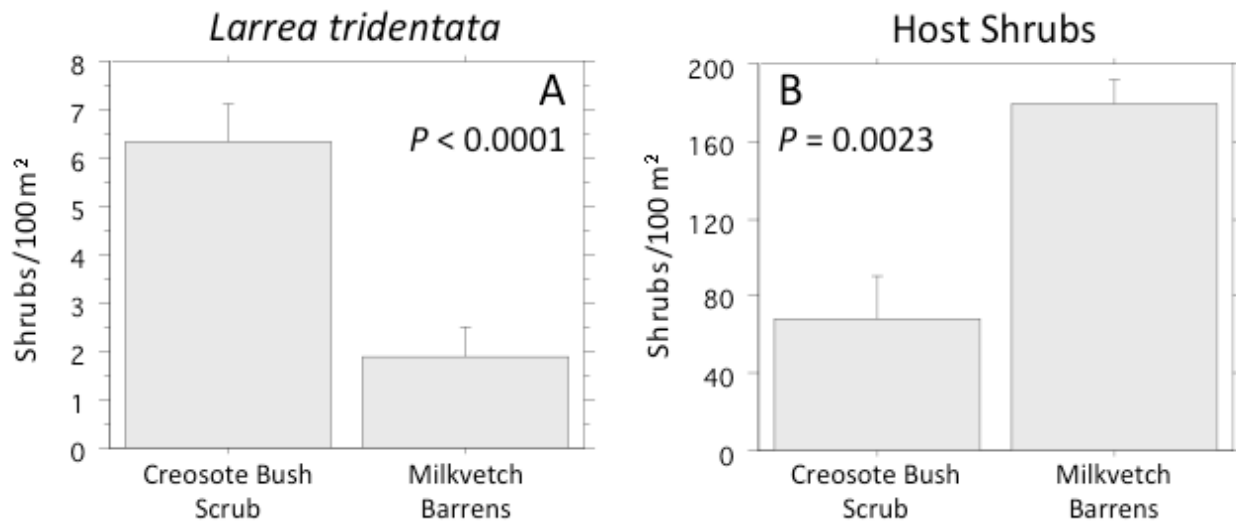


Figure 5. Differences in the distribution of *Larrea tridentata* (A) and potential *Astragalus jaegerianus* host shrubs (B) in creosote bush scrub (CBS) and milkvetch granite barrens (MGB). *Larrea tridentata* is significantly more abundant in CBS than MGB, and potential *A. jaegerianus* host shrubs are significantly more abundant in MGB than CBS. Bars are means (+ SE).

Drought Effects

Three of the five paired sites that we surveyed in 1999 were resurveyed in 2010. The effects of drought on the vegetation structure in the last 10 years are presented in figure 7. Total shrub mortality (166 shrubs) was high compared to shrub recruitment (16 shrubs), and the majority of mortality and recruitment occurred in milkvetch communities (131 deaths and 16 recruits). Values for density and cover were lower in 2010 than in 2000, but the mean shrub density within the granite barrens remained twice as high as pre-drought shrub densities in creosote bush scrub.

Decreases in density represent the death without replacement of shrubs, and decreases in cover represent both death of shrubs and drought pruning of live shrubs. Within shrub transects, shrubs species responded differently to the drought as some species are capable of surviving drought better than others. Within milkvetch barrens, only *Larrea tridentata* maintained the same density over the 10 year period, and the hemi-parasite *Krameria erecta* increased in density. Most species declined 20–50 percent, and *Ericameria cooperi* and *Lycium andersonii* A. Gray, *L. cf. cooperi* A. Gray (Anderson's desert thorn) declined

over 75 percent in density. Species that were little affected by the drought include *Larrea tridentata*, *Thamnosma montana*, *Ephedra nevadensis*, *Salazaria mexicana*. These species maintained 80 percent or more of their measured density in 2000. Within creosote bush scrub sites, *Larrea tridentata* and *Krameria erecta* responded similarly to the drought as they did at the milkvetch barren sites; other species that are found at both sites (*Ambrosia dumosa*, *Ephedra nevadensis*, and *Thamnosma montana*) had slightly greater declines than observed on the milkvetch barrens, and species more-or-less unique to the creosote bush scrub sites (*Lycium andersonii*, *Krascheninnikovia lanata* (Pursh) A. D. J.

Meeuse & Smit) (winterfat) suffered big declines in density.

Astragalus jaegerianus mortality was high during the drought at our long-term study sites: At Brinkman Wash, only 13 of 109 *A. jaegerianus* survived from 1999, and at the Gemini Conservation Area only 7 of 43 *A. jaegerianus* survived from 2003 (table 1). The highest *A. jaegerianus* mortality (19 plants) occurred in *Eriogonum fasciculatum*, the most common host shrub in 1999 and in 2003 with 41 *A. jaegerianus*. By 2010, *Thamnosma montana* Torr. & Frem. was the most common host shrub with 5 *A. jaegerianus*, followed by *Ambrosia dumosa* (4), *Ephedra nevadensis* (3), and *Eriogonum fasciculatum* (3).

A. All shrub species

Species	Observed	Expected	(O-E) ² /E
<i>Eriogonum fasciculatum</i>	22	15.69	2.53
<i>Larrea tridentata</i>	1	13.90	11.97
<i>Ericameria cooperi</i>	7	10.98	1.44
<i>Ephedra nevadensis</i>	15	9.16	3.71
Other species	27	22.25	1.01
	72		20.66 **

B. All shrub species excluding *Larrea tridentata*

Species	Observed	Expected	(O-E) ² /E
<i>Eriogonum fasciculatum</i>	22	18.31	0.74
<i>Ericameria cooperi</i>	7	13.87	3.40
<i>Ephedra nevadensis</i>	15	10.58	1.84
<i>Ambrosia dumosa</i>	6	5.94	0.00
<i>Krameria erecta</i>	1	5.29	3.47
Other species	20	17.01	0.53
	71		9.98

C. *Larrea tridentata* versus all other shrub species

Species	Observed	Expected	(O-E) ² /E
<i>Larrea tridentata</i>	1	13.905	11.977
other species	71	58.095	2.867
	72		14.844 **

Table 3. *Astragalus jaegerianus* host shrub preference using chi-square analysis with (A) creosote bush (*Larrea tridentata*), (B) without creosote bush, and (C) creosote bush versus all other shrubs. Creosote bush is significantly under-represented as a *A. jaegerianus* host shrub, and is probably unsuitable (n = 72 total host shrubs). "****" = significant (P = 0.01).

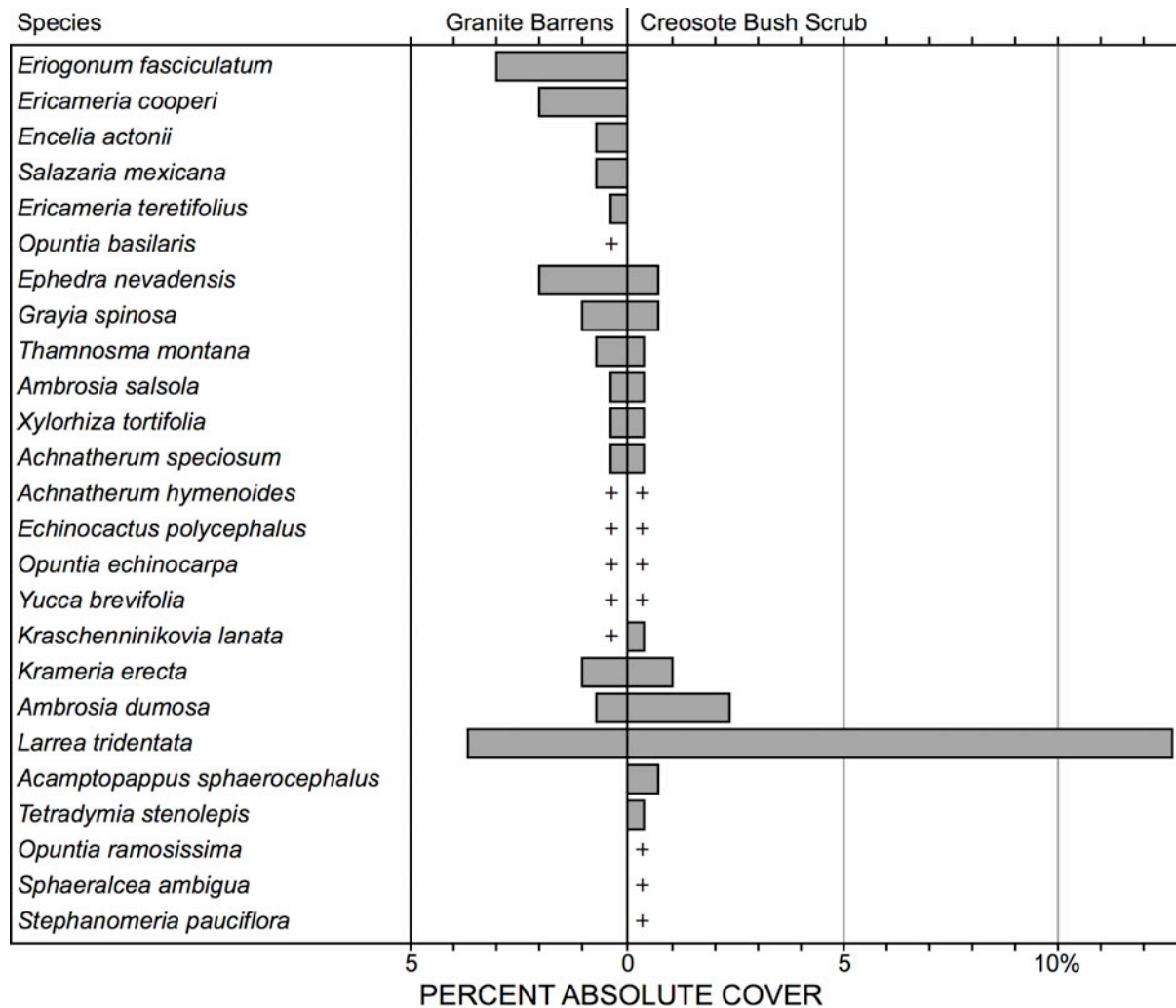


Figure 6. Mean absolute (percent) cover for perennial shrubs occurring within belt transects in granite barrens and adjacent creosote bush-dominated vegetation, at the National Training Center, Fort Irwin, CA (2000). Bars are mean absolute cover of five study sites rounded to the nearest 1/3 percent; “+” = plants with < 0.1 percent cover.

DISCUSSION

Granite barrens of the central Mojave Desert (900 to 1200 m) support shrub communities derived from the dominant, surrounding vegetation of the area, but modified by edaphic circumstances into distinctive communities that harbor the endemic species, *Astragalus jaegerianus*. The vegetation that surrounds the granite barrens is a transition vegetation composed of lower elevation creosote bush scrub (below sea level to 1400 m) with elements of higher elevation Joshua tree woodlands (1055 to 1525 m) (Thorne 1982). The shrub flora of granite barrens is derived entirely from these two plant communities, but the relative abundance, density, and size of shrubs is both visually and statistically

different: the shrub community on granite barrens is composed of smaller shrubs at a significantly higher density than in the surrounding creosote bush dominant community, and three of the most common shrubs on granite barrens, *Eriogonum fasciculatum*, *Ericameria cooperi*, and *Ephedra nevadensis*, are a minor component of the adjacent creosote bush dominant community. The six species found exclusively in granite barren transects (figure 6) are characteristic of a subclass of desert rupicolous scrub described by Thorne (1982) as “mixed desert scrub”: an open, edaphically controlled community found on rocky, non-calcareous slopes where soil development is restricted. Mixed desert scrub is one of the most complex and least understood desert plant communities because it varies floristically with

latitude, rainfall, and substrate, and was thought by Thorne to involve more than one community (Thorne 1982). Like mixed desert scrub, the perennial flora of granite barrens lacks clearly dominant species, but displays an impressive variety of desert growth habits including shrubby species (for example, *Eriogonum fasciculatum*, *Ericameria cooperi*, *Ephedra nevadensis*, *Larrea tridentata*), annuals (for example, *Phacelia tanacetifolia* Benth., *Pholistoma auritum* (Lindley) Lilja, *Coryopsis biglowvii* (A. Gray) H. M. Hall) (Bigelow's coreopsis), stem-succulents (*Opuntia*), leaf semi-succulents (*Yucca*), perennial grasses (*Achnathrum speciosum* (Trin. & Rupr.) Barkworth (desert stipa), and herbaceous perennials such as *Astragalus jaegerianus*. As a consequence of (1) the unique properties of the shrub community on granite barrens relative to the surrounding dominant vegetation, (2) the limited geographic distribution of these barrens in the central Mojave Desert, (3) the prominent role of edaphics in controlling community structure, and (4) the presence of the endemic indicator species *Astragalus jaegerianus*, a community level designation "milkvetch barrens" as a distinct sub-community within Thorne's mixed desert scrub is proposed here.

Soil chemistry is unlikely to be the edaphic factor responsible for the stunted vegetation of milkvetch barrens: while some difference in soil chemistry exist between milkvetch and adjacent creosote bush sites (table 2), these differences are typically small and fit

easily within the range of Mojave Desert soils that support creosote bush scrub communities (Graham and O'Geen 2010; Cox and others 1984; Romney and others 1973; Schlesinger and others 1996; Rundel and Gibson 1996). While the soil of milkvetch barrens tends to be marginally less fertile than adjacent creosote bush scrub (in other words, higher N and P, table 2), these differences are unlikely to be the source of vegetation differences because other Mojave Deserts soils are less fertile than milkvetch granite barrens but support creosote bush scrub communities (Schlesinger and others 1996). A more likely edaphic agent for the stunting of the vegetation of milkvetch barrens is soil moisture. Milkvetch barrens and adjacent creosote bush-dominated sites share the same abiotic conditions of precipitation and temperature, but differ appreciably in soil depth; 0 to 40 cm in milkvetch barrens versus greater than 100 cm in adjacent, creosote bush-dominated sites. Since the texture of soils in milkvetch and creosote bush sites are nearly identical (table 2), the shallow soils of milkvetch barrens are likely to have less total water holding capacity relative to adjacent deep soil creosote bush sites, resulting in a reduction in shrub cover and other plant metrics (figure 3). Previous studies of edaphic communities have reported decreases in soil moisture as soils become shallower, with a resulting decrease in plant cover (Baskin and others 1972; Sharitz and McCormick 1973; Meyer 1986).

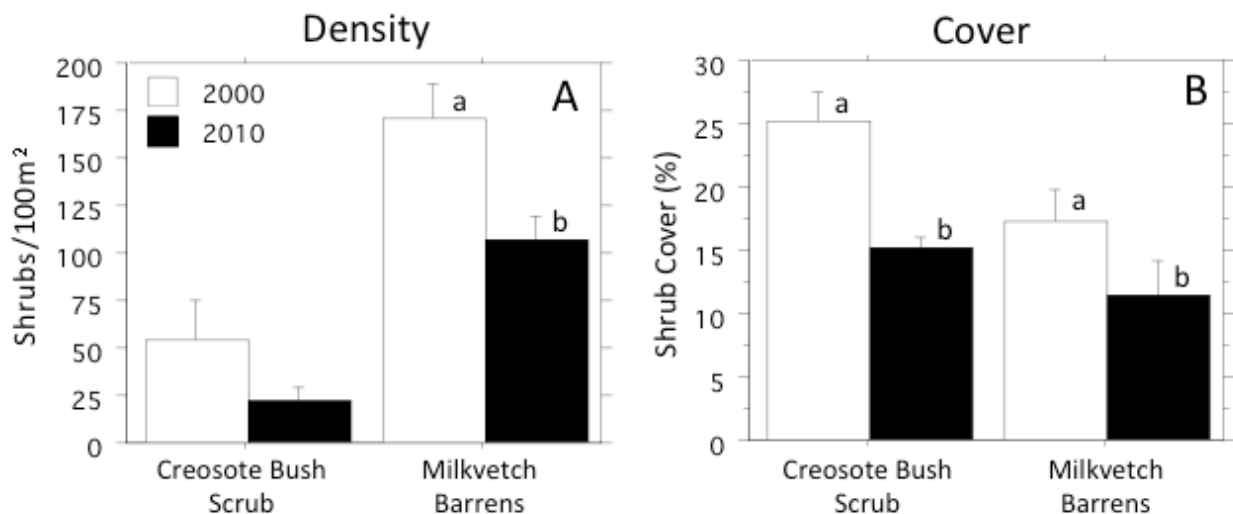


Figure 7. The effect of drought (1999-2010) on shrub density (A) and shrub cover (B) between milkvetch barrens and creosote bush scrub communities. Between 1999 and 2010, each community experiences declines in shrub density and cover, some of which were significant (*): milkvetch barrens density, $P = 0.0216$; milkvetch barrens cover, $P = 0.0029$ creosote bush scrub cover, $P = 0.0264$. Bars are means (+ SE).

Reductions in shrub cover on milkvetch barrens could occur in a number of ways (figure 8): through decreases in shrub size (figure 8B), through decreases in shrub density (figure 8C), or through some combination of change in both size and density of shrubs (figure 8D). In the case of milkvetch barrens, this reduction in cover has occurred through a proportional reduction in shrub size and inter-shrub distance (figure 8D). Consequently, on milkvetch barrens, shrub density increases and inter-shrub distance decreases relative to adjacent, creosote-dominated communities on deep alluvial soils.

Our previous seed bank studies have established that *A. jaegerianus* is a poor disperser, such that the dispersal of seed beyond host shrubs with seed-producing *A. jaegerianus* is extremely rare, and limited to near-by shrubs (Rundel and others 2009). High host shrub densities on milkvetch barrens (and consequent decreases in inter-shrub distance) could increase the likelihood of *A. jaegerianus* seed dispersal to other host shrubs, and as a result, restrict it to milkvetch barrens, where the probability of colonizing new shrubs is higher than in surrounding creosote bush-dominated communities. Hypothetically, low host shrub densities in creosote bush-dominated communities could create inter-shrub distances too great to support *A. jaegerianus* dispersal, effectively blocking expansion of *A. jaegerianus* into these areas. If *A. jaegerianus* population growth is limited by dispersal, and dispersal increases with increased shrub density, the drought-induced decreases in host shrub density observed since 1999 (figure 7A) may be a contributing factor in the failure of *A. jaegerianus* to recruit new plants in 11 years of observation.

Astragalus jaegerianus is a novel example of a second-order edaphic endemic whose distribution is indirectly controlled by edaphics through the effect of edaphics on its community of host shrubs. This indirect effect of edaphics is analogous to that of the cedar glade endemic *A. tennesseensis* A. Gray (Baskin and others 1972), in which shallow, rocky soils modify and reduce the dominant vegetation type into suitable habitat for *A. tennesseensis*. The degree to which indirect effects of edaphics on *A. jaegerianus* conform to models of edaphic endemism is unclear. On one level, *Astragalus jaegerianus* appears to conform to Gankin and Majors' (1964) "refuge model", in that *A. jaegerianus* is restricted to the shallow soils of milkvetch barrens that exclude the dominant creosote bush community because of a disadvantage:

insufficient soil moisture. Nevertheless, the shallow soils of milkvetch barrens are not an unambiguous "disadvantage" to *A. jaegerianus*, and so *A. jaegerianus* also appears to conform Meyer's (1986) "specialist" model, in that *A. jaegerianus* "is excluded from adjacent soils because its adaptation to its own soil has rendered it less able to survive on other types of soil." However, in the case *A. jaegerianus*, specialization does not occur in relation to the edaphics of granite barrens, but rather, to the shrub community that the edaphics of granite barrens produce. Ultimately, *A. jaegerianus* may not fit either the refuge or specialist model, and may require a new "indirect model" of edaphic endemism to explain its presence on granite barrens in central Mojave Desert.

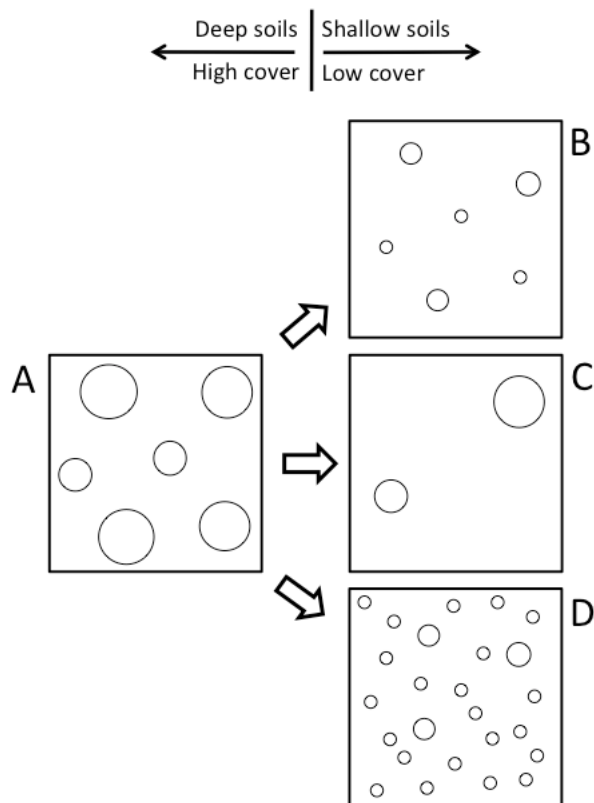


Figure 8. Diagram showing alternative edaphic-controlled transitions from a high-cover shrub community like creosote bush scrub, to low cover shrub community like that of milkvetch barrens in which shrub size has decreased (B), shrub density has decreased (C), or both shrub size and inter-shrub distance have decreased but the size and distance relationships between shrubs are maintained (D). Both community B and C are reasonable, *a priori* models of the effect of decreased soil moisture on community A, but community D more closely resembles the structure and organization of shrub communities on "milkvetch barrens".

CONCLUSIONS

1. The vegetation of "milkvetch barrens" in the central Mojave Desert (900 to 1200 m) is a desert rupicolous scrub that should be considered a distinct sub-community within Thorne's "mixed desert scrub" because of its distinctive edaphically controlled vegetation structure, its limited distribution, and the presence of the edaphic endemic *A. jaegerianus*.
2. Results suggest that *A. jaegerianus* does not show a preference for host shrub, except that *Larrea tridentata* is significantly under-represented as a host shrub and so appears to be an unsuitable host shrub for *A. jaegerianus*. This unsuitability may be due to *Larrea*'s architecture, which is significantly taller than suitable host shrubs, and too tall and open to allow *A. jaegerianus* to climb through and reach its canopy.
3. Drought condition since 1999 have led to decreases in density and cover in both milkvetch barrens and adjacent creosote bush-dominated communities, but the density of host shrubs in milkvetch barrens in 2010 remain twice that of pre-drought creosote bush-dominated communities. Drought-induced decreases in host shrub density may be a contributing factor in the failure of *A. jaegerianus* to recruit new plants since 1999.
4. The shrub community on milkvetch barrens is composed of smaller shrubs at a significantly higher density than in the surrounding creosote bush dominant community. Soil depth is likely to be the edaphic factor responsible for the stunted vegetation of milkvetch barrens, because their shallow soils have less total water holding capacity relative to adjacent deep-soil creosote bush sites, resulting in a reduction in shrub cover and other plant metrics.
5. The higher shrub density of milkvetch barrens reduces inter-shrub distance, which is hypothesized to increase *A. jaegerianus* dispersal and population growth. Reciprocally, low host shrub densities in adjacent creosote bush-dominated communities could create inter-shrub distances too great to support *A. jaegerianus* dispersal, effectively blocking expansion of *A. jaegerianus* into these areas.

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Synergistic Monitoring – Addressing the Threats and Identifying Opportunities

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ABSTRACT

For many years, land managers and scientists have been applying a variety of land treatments to improve or protect rangeland ecosystems. Collectively, we have studied the response of these treatments and wildfire events to identify opportunities for maintaining or improving Nevada sagebrush ecosystem health and functionality. In partnership with collaborators, we initiated a State-wide effort to capture, consolidate, and summarize implementation, monitoring, and research information for these events. We are conducting field studies to identify and fill information gaps. We seek a new and expanded information base that is available to Nevada land managers, scientists, and others interested in healthy and resilient sagebrush sites. We plan to identify the consequences of passive and active management; develop predictive tools for adaptive management; identify research needs; and increase accessibility to location, implementation and monitoring information for these events. Through the collaborative integration of our field study results with historic and current research and monitoring information, we seek to increase knowledge of landscape-level and site-specific ecological processes. This will further develop our ability to manage and predict rangeland health, integrity, resilience (after disturbance), and resistance (to undesired change under significant disturbance regimes) in the context of multiple-use management.

In Monaco, T.A. et al. comps. 2011. Proceedings – Threats to Shrubland Ecosystem Integrity; 2010 May 18-20; Logan, UT. Natural Resources and Environmental Issues, Volume XVII. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan Utah, USA.

INTRODUCTION

We estimate that more than 25,000 land treatment and wildfire events have occurred in Nevada since the early 1900's, and land managers and scientists from across the United States have conducted substantial monitoring and research studies on many of them (Swanson and others 2010). In 2008, the Synergistic Monitoring Project (SynMon) initiated the collaborative harvesting and compilation of implementation, monitoring, and research information available for these events. The purposes of this effort are: (1) to capture and summarize what is currently known about event outcomes; (2) to facilitate and support future study of established monitoring and research sites; and (3) to identify information gaps that we plan to bridge through follow-on field studies and data publication. The intent is that all of the activities described below will occur each year over the project's lifetime, according to annual geographic,

ecologic, and/or other topical focus areas developed in conjunction with our collaborators. We intend to widely share this information for its future use in the study and management of wildfires and land treatments across Nevada's sagebrush ecosystems.

Information Harvesting

Through the generous support of collaborators, we currently have a spreadsheet populated with varying amounts of information for each of over 6,000 wildfire, land treatment, research, and related Nevada events. We also house a database containing location, implementation, planning, and/or monitoring or research information for many of these sites. We continue to add more information for listed events and new events as time and funding permit and new information becomes available.

It is important to note that the USGS's Great Basin Integrated Landscape Monitoring Pilot (GBILMP) Project has a similar information harvesting and analysis effort underway for those Great Basin lands under the jurisdiction of the U.S. Department of Interior (USDI 2007). GBILMP and SynMon have been mutually supportive, although our SynMon project has also been harvesting information from U.S. Department of Agriculture (U.S. Forest Service, Natural Resources Conservation Service, and Agricultural Research Service) and several state, county and private management, academic, and other entity offices.

In the long term, we hope to build a complete, easily-accessed spreadsheet and database containing the locations, implementation data, and ecological outcomes for all Nevada sagebrush ecosystem wildfire and land treatment events. In the interim, we are using currently stored information to help identify information needs, to build plans for out-year monitoring and research activities, and to centrally place-hold key monitoring and research information related to Nevada sagebrush wildfires and land treatments.

Identification of Information Needs

During 1999-2006, almost 6 million acres of Nevada lands experienced wildfire, with some sites burning multiple times (Kozlowski and others 2010). Plant communities on many of these lands subsequently transitioned from native vegetation states into cheatgrass *Bromus tectorum* L. and/or exotic forb-dominated states (Miller and Narayanan 2008).

Through the course of several collaborator meetings, it became clear that northeastern Nevada was a priority area for which many ecological questions exist. Within this geographic area, wildfire and land treatment questions focused on Wyoming big sagebrush sites. A need was expressed for learning about ecological resiliency and resistance threats as well as opportunities posed by wildfires and various land treatments under varying ecological site and state scenarios. Interest was shown in the identification of threshold points, interactions among shrubs and herbaceous vegetation, and decision tools

for wildfire and land treatment management. Also, we discussed the identification of locations and foci for future research and the potential for investigating lower-intensity field data collection techniques covering multiple parameters and applicable across larger geographical scales. Field studies were planned accordingly.

Field Studies

In 2010, we completed reconnaissance-level field studies on 50 northeastern Nevada wildfire, preventative land treatment (aerating, burning, disking, herbicide spraying, mowing), and aroga moth visit sites. For field study purposes, we defined preventative land treatments as those native vegetation manipulation efforts designed to directly or indirectly protect, maintain, or improve native plant community health, functionality, diversity, resiliency, and/or resistance to invasive species occupation under wildfire or other significant disturbance events. None of the studied sites were known to have been seeded, or had unwanted vegetation control or other rehabilitative kinds of treatments as part of the respective event. The purpose of these studies was to specifically address the following:

1. A scientist contemplating a new land treatment for study might be challenged by finding a place to do the treatment, getting the land owner to permit the treatment, paying for the treatment, and of course, getting the treatment completed. Can similar experimental quality be achieved by studying existing treatments that are already in place?
2. Do low-intensity, reconnaissance-level data collection techniques provide data sensitive enough to effectively support the analysis of targeted ecological parameters at larger scales? Do they quantify apparent outcome differences - such as those apparent in Figures 1 and 2?
3. Can the study sites be designed and located in a manner such that others may easily locate and re-study them (or integrate them with other studies) in the future?



Figure 1. A northeastern Nevada Wyoming big sagebrush site that experienced wildfire in 2006. Elevation: 5630 feet. Vegetation dominants: cheatgrass and exotic forbs.



Figure 2. Another northeastern Nevada Wyoming big sagebrush site that experienced wildfire in 2006. Elevation: 5360 feet. Vegetation dominants: native perennial graminoids, mostly bottlebrush squirreltail, *Elymus elymoides* (Raf.) Swezey.

4. Address several hypotheses:

- a) Certain events lead to vegetation responses that are more likely to result in perennial resilience and long-lasting fuel reduction, wildlife habitat, livestock forage, and other benefits.
- b) The outcome of events differs among ecological sites.

c) Vegetation response differs markedly as the pre-event cover of shrubs exceeds some proportion of total plant cover in pre-event vegetation composition or in non-event control sites.

d) At levels of shrub cover close to the threshold proportion, event response is strongly correlated with the proportion of annual versus perennial herbaceous species cover in the pre-event or control site shrub understory.

e) Event responses differ according to several factors, such as land use or general management practices; weed infestations; topography or elevation; duration of fire exclusion period; and/or treatment implementation characteristics.

f) The effects become obvious within five years and remain obvious for at least two additional decades after the event (some locations will not allow testing of this hypothesis for some time).

The point intercept technique was used, since it can minimize observer bias and inter-observer variation among years (Wirth and Pyke 2007), and can capture a variety of abiotic and biotic structural component parameters. In an effort to complete data collection on as many sites as possible, we used a minimal sampling intensity - 200 points per event site, and 200 points for each adjacent control site – which Mueller-Dombois and Ellenberg (1974) indicate may yield satisfactory results. We collected both ground-level (bare soil, litter, rock, cryptogam, and basal vegetation) and vegetation foliar (live, dead, and decadent) cover by species on event and control areas. Shrub canopy height and width dimension data were also collected. All sites were mapped, benchmarked, and photographed, per Perryman and others (2006) and Swanson and others (2006).

We have initiated the analysis of these field data and will publish results as sufficient data permit testing of one or more hypotheses. This should continue in other geographic areas until the conclusion of this effort.

APPLICATIONS

We anticipate that the subsequent integration of SynMon and other monitoring and research data from northeastern Nevada Wyoming Big Sagebrush sites will contribute knowledge toward:

1. Identification of those ecological sites and/or states in which a particular wildfire could be managed for its beneficial effects, versus those for which a wildfire should be extinguished to avoid detrimental effects.
2. Identification of the kinds of land treatments that might best meet management objectives under a variety of scenarios.
3. The scheduling and/or programming of out-year maintenance treatments.
4. Conducting environmental effects and other analyses – such as efforts directed toward comparing the effects of alternative land treatments and no treatment.
5. Achieving and sustaining ecological health, functionality, and resiliency.
6. Permanent benchmarking of historic and current monitoring and research sites.
7. Streamlined monitoring protocols to address multiple information needs across larger scales.

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Moderating Livestock Grazing Effects on Plant Productivity, Nitrogen and Carbon Storage

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ABSTRACT

Multi-year studies of plant communities and soils in the Bear River Range in southeastern Idaho and northeastern Utah found reduced ground cover and herbaceous production in areas grazed by livestock when compared to reference values or long-term rested areas. Reductions in these ecosystem components have led to accelerated erosion and losses in stored carbon and nitrogen. Restoration of these ecosystem components, with their associated carbon and nitrogen storage, is possible by application of science-based grazing management.

In Monaco, T.A. et al. comps. 2011. Proceedings – Threats to Shrubland Ecosystem Integrity; 2010 May 18-20; Logan, UT. Natural Resources and Environmental Issues, Volume XVII. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan Utah, USA.

INTRODUCTION

During the past two decades, the role of carbon emissions in climate change has heightened interest in carbon sequestration as a means of mitigating climate change (FAO 2009). Forests sequester 86 percent of the planet's above-ground carbon and 73 percent of the planet's soil carbon (Sedjo 1993). Studies conducted on the Wasatch-Cache National Forest in the Bear River Range in Idaho and Utah found that ground cover and herbaceous vegetation production were reduced at sites grazed by livestock when compared with sites that had been rested for long periods or with reference values. Additionally, the loss of ground cover in grazed areas has led to accelerated soil erosion. Studies of soil organic matter, carbon, and nitrogen were conducted since erosion of the surface soils could have resulted in loss of these constituents or displacement to other locales, where mineralization could be increased by greater exposure to oxygen, light and water. For example, carbon losses from soil erosion can occur by reductions in soil productivity in the eroding areas (Schuman and others 2002).

Worldwide, soil organic matter contains three times as much carbon as the atmosphere (Allmaras and others 2000; ESA 2000; Flynn and others 1960). Rangelands have been estimated to store 30 percent of the world's soil carbon with additional amounts stored in the associated vegetation (Grace and others 2006; White and others 2000). Past rangeland use in the United States has led to losses in soil carbon

(FAO 2009; Follett and others 2001). It is estimated that 73 percent of rangelands worldwide have suffered soil degradation (WOCAT 2009). This is significant in the eleven western states (Arizona, California, Colorado, Idaho, Montana, New Mexico, Nevada, Oregon, Utah, Washington, and Wyoming), where 305,000,000 acres of public land (National Forests, National Parks, National Wildlife Refuges, Bureau of Land Management, state and county lands) are leased for grazing livestock. An additional 220,000,000 acres of Indian reservations and private lands in these states are also grazed by livestock (Wuerthner and Matteson 2002).

Soil organic carbon is an important source of energy that drives many nutrient cycles. Increases in soil organic matter lead to greater pore spaces and more soil particle surface area which retains more water and nutrients (Tisdale and others 1985). Soil organic carbon, which makes up about 50 percent of soil organic matter, is correlated with soil fertility, stability, and productivity (Herrick and Wander 1998). Soil organic carbon and nitrogen decline in concentration from surface to subsoil with the highest rates of mineralization activity occurring in the top 2.5 cm of soil and beneath vegetation (Charley and West 1977; Yang and others 2010).

The loss of topsoil as a result of accelerated erosion resulting from livestock grazing has been well documented and affects these more organic and nutrient-rich surface layers first. Livestock grazing can compact the soil, reduce infiltration, and increase

runoff, erosion, and sediment yield (Ellison 1960; Warren and others 1985). White and others (1983) found that sediment yield was 20-fold higher in a grazed watershed than in an un-grazed watershed. Numerous studies have observed severe erosion in the western United States when comparing heavily grazed areas to un-grazed areas (Cottam and Evans 1945; Gardner 1950; Kauffman and others 1983; Lusby 1979). There are also a number of extensive literature reviews on this topic that describe the impact of livestock grazing on soil stability and erosion (Fleischner 1994; Gifford and Hawkins 1978; Jones 2000). Removal of plant biomass and lowered production resulting from livestock grazing can reduce soil fertility and organic matter content (Trimble and Mendel 1995).

The grazing of livestock accelerates the rate of conversion of vegetation to gaseous forms of emissions. West (1983) noted that grazing and fire serve to accelerate the recycling of ash elements and result in gaseous losses of nitrogen. West (1981) noted that nearly all the nitrogen returned in animal feces and urine is lost in gaseous forms. Worldwide, livestock production accounts for about 37 percent of global anthropogenic methane emissions and 65 percent of anthropogenic nitrous oxide emissions. Methane emissions from cattle range from 6 to 7 percent of forage consumed (FAO 2006).

METHODS

Study Area

The Bear River Range occurs in the Caribou-Targhee National Forest in Idaho and in the Uinta-Wasatch-Cache National Forest in Utah (figure 1). The Utah portion contains 28 allotments that are grazed by cattle or sheep. The portion of the Bear River Range in Idaho contains 26 allotments grazed by cattle or sheep. Livestock have grazed here since the late 1800's. Grazing management has relied on increasing the number of water developments or rotation grazing systems. Authorized utilization levels range up to 55 percent (USDA 2004; USDA 2005). In the North Rich allotment, where the production and soil chemistry data were collected, a three pasture rest-rotation grazing system was implemented in the 1970's and abandoned a few years later due to fence maintenance issues. In a 2004 Forest Service decision (USDA 2004), the system was reinstated, but has not yet been implemented on the ground. The

permitted stocking rate has remained essentially unchanged with season-long grazing since the 1960's with some year-to-year variation based on drought or permittee needs and the short-term implementation of the rest-rotation system.

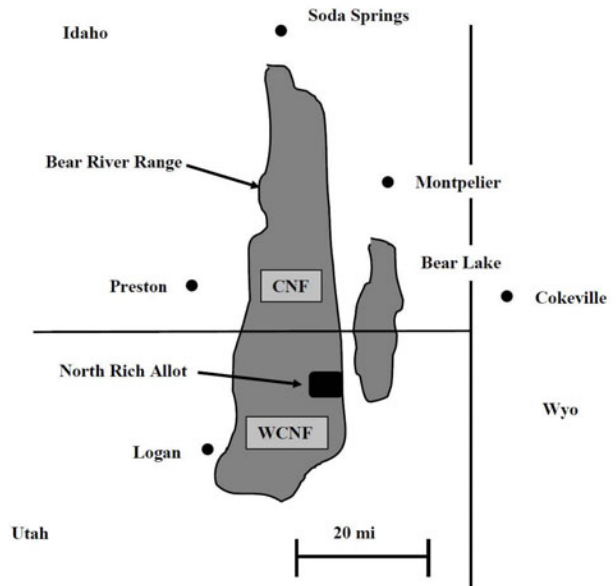


Figure 1. Map of Study Area.

Elevations range from 5,000 to near 10,000 feet with precipitation ranging from 12 inches at lower elevations to 40 inches at the higher elevations. Geology is a combination of karst and sedimentary types with dominant vegetation consisting of Douglas fir, mixed-conifer, aspen, mountain big sagebrush and mountain brush. Topography is steep with narrow valley bottoms and large, open basins on the crest of the range with rolling foothills in sagebrush-steppe at lower elevations (USDA 2003a; USDA 2003b). Under constant grazing pressure by livestock, plant communities have been altered with sensitive native bunchgrasses being replaced by more grazing-tolerant grasses and forbs being dominated by less palatable species.

Ground Cover Studies

Ground cover and soil surveys were conducted during the period 1990 to 2008. These were initially focused on two allotments in the Utah portion of the Bear River Range and then were expanded in 2001 to include locations in the Idaho portion of the Range. Locations were selected in mountain big sagebrush, aspen or mixed-conifer representative of lands accessible to cattle with gentle slopes and available

water. Two methods of measuring ground cover were used. A rapid assessment method using a 34-inch diameter hoop placed at 10 yard intervals along a 100 yard transect was used for most data collection due to the large number of sites measured. Ground (basal) cover of grasses, forbs, shrubs, litter, rock, crust and bare ground were estimated to the nearest 1 percent. Ocular estimates were calibrated using a standard area card that was 1 percent of the plot area for comparison. A second method employed a nested frequency frame to collect more intensive data and to validate the ocular estimates. Five 100-foot radial transects were oriented from the center point. Along each transect, a nested frequency frame with eight points was placed at five foot intervals, recording ground cover "hits" for each point. A total of 800 points were recorded for each site surveyed. Sites surveyed by both methods gave similar results. Time savings by using the ocular method were significant in that the ocular method took approximately 30 minutes at a site, while the nested frequency method took over two hours at a site.

Production Studies

Herbaceous (grass and forb) production was determined by clipping plots at each site. Plots were clipped in the North Rich allotment in 2001 (five plots per site), 2004 (three plots per site) and 2005 to 2007 (one caged plot per site). The 2001 and 2004 plots were clipped prior to livestock entry into the allotment. The 2005 to 2007 plots were clipped after the grazing season. These plots were protected inside utilization cages and represent un-grazed samples. Adjustments for plant phenology were applied to the plot data from the 2001 and 2004 samples. Post-grazing samples needed no phenology adjustments since the growing season was complete prior to sampling. A 36" x 36" plot frame was used. Samples were clipped to 1/2" above the soil surface, placed in Ziploc™ bags and returned to the office for air drying. Samples were initially air dried to a constant weight in a warm space at about 80° F resulting in the clipped samples being brittle and easily broken. Subsequent samples were air dried to this textural endpoint. Based on oven-drying of subsamples, the air-dried samples contained about 5 percent moisture. Once dry, samples were weighed on an electronic balance sensitive to 0.1 gram.

Soil Studies

Soil samples were taken of the top 4 inches below the litter layer. Triplicate subsamples were taken at each ground cover transect location and combined. These were placed in bags and kept in a cooler with ice until delivered to the Utah State University Analytical Laboratory in Logan, Utah. Methods of analysis included determination of soil organic matter by loss on ignition, total organic carbon by the combustion method and total nitrogen by the Kjeldahl method (Miller and others 1997). Soil pits were excavated at each site and inspected for root density, soil stability and organic matter.

RESULTS

Literature - Ground Cover and Production Data

A search of agency records was used to determine the potential and historical basal ground cover of grasses, forbs, shrubs, litter, rock, biological crust, and herbaceous production for the plant communities of interest including aspen, conifer and mountain big sagebrush. These are presented and used for comparison with the results of surveys for ground cover, herbaceous production and soil chemistry recently conducted in the Bear River Range.

Ground Cover

The Wasatch-Cache National Forest (WCNF) provided data from nested frequency transects considered representative of potential ground cover (USDA 1996). Potential ground cover values for aspen ranged from 90 to 98 percent and mountain big sagebrush ranged from 81 to 96 percent (table 1). There were no potentials given for mixed-conifer forest. Caribou National Forest (CNF) plot data for percent bare soil (average of maximum values, average of all values, and average of minimum values) were summarized from historical range analyses (1959 to 1976) for the Montpelier Ranger District, which includes the Bear River Range (USDA 1997). The maximum ground cover values found in those range analyses are consistent with the highest values used as reference in the WCNF and appear to represent potential values of 98 percent for aspen, 94 percent for mountain big sagebrush and 98 percent for mixed conifer (table 1).

Table 1. Forest Service ground cover determinations, percent.

Source/Vegetation Type	Aspen	Mountain Big Sagebrush	Conifer
Reference or Potential Values – Ungrazed Areas			
WCNF (USDA 1996)	90 – 98	81 – 96 ^a	--
USDA (1997)	98	94	98
Grazed Areas			
WCNF (1962)	79	59/70 ^b	75
WCNF (USDA 2004)	67	36 – 87	--
CNF (USDA 1997)	98/62/85 ^c	94/39/70	98/67/87

^aIncludes Silver sagebrush (*Artemisia cana*) and mountain big sagebrush (*Artemisia tridentata vaseyana*).

^bHinger number is from “unsuitable” lands that received lower grazing intensity due to slope or distance to water.

^cMaximum / minimum / average from CNF range analysis plots. Maximum and minimum plots averaged from all locations. Average is the average for all plots across all locations.

Ground cover conditions in these plant communities during Forest Service range analysis surveys in areas grazed by livestock were well below these potentials (table 1). Ground cover in aspen communities in the North Rich allotment (1961 and 1962) was 79 percent, while mountain big sagebrush was 59 percent and conifer was 75 percent. Historical data for sites in areas grazed by livestock in the CNF averaged 85 percent ground cover in aspen, 70 percent in mountain big sagebrush, and 87 percent in conifer sites, with much lower values at many sites. Range analysis data for the North Rich allotment from the 1970's for sagebrush, meadow and aspen communities, averaged across all sites, had average ground cover of 56 percent (USDA 1989). WCNF data collected in 2002 for the North Rich allotment found 67 percent ground cover in aspen and 36 to 87 percent in mountain big sagebrush (USDA 2004).

Production

Potential herbaceous plant community production values (table 2) were taken from Natural Resources Conservation Service (NRCS) ecological site descriptions that best matched the aspen and mountain big sagebrush sites surveyed (USDA 1992a,b,c). Based on these descriptions, during average precipitation years, mountain stony loam aspen communities produce 2,150 to 2,250 lbs/acre consisting of 45 percent grass and 30 percent forbs. Mountain loam mountain big sagebrush communities produce 1,600 to 2,000 lb/acre with 80 percent grasses and 5 percent forbs. Mountain shallow loam mountain big sagebrush communities produce 1,000 to 1,100 lb/acre with 50 percent grass and 5 percent forbs. No ecological site descriptions directly applicable to the mixed-conifer were found.

Given the maximum, average, and minimum production values published by the CNF (USDA 1997) and the average grass and forb percentages from the source data (table 2), herbaceous production was calculated for the Caribou National Forest. Based on this calculation, aspen communities produced a maximum of 1,297 lb/acre per year with an average of 654 lb/acre and a minimum of 297 lb/acre. Mountain big sagebrush communities produced a maximum of 914 lb/acre per year with an average of 453 lb/acre and a minimum of 153 lb/acre. Conifer communities produced a maximum of 780 lb/acre per year with an average of 348 lb/acre and a minimum of 107 lb/acre. Historical data from the 1961 and 1962 range analyses for the North Rich allotment found that aspen communities produced 241 lb/acre grasses and 443 lb/acre forbs for a total herbaceous production of 684 lb/acre. Mountain big sagebrush communities produced 122 lb/acre of grass and 163 lb/acre of forbs for a total herbaceous production of 285 lb/acre. Mixed-conifer communities produced 157 lb/acre grass and 253 lb/acre forbs for a total herbaceous production of 410 lb/acre.

Ground Cover Surveys

Surveys of ground cover conditions were conducted throughout the Bear River Range in Idaho and Utah (table 3). In 2001, 41 grazed and three un-grazed mountain big sagebrush locations were surveyed in the CNF. Mountain big sagebrush locations grazed by livestock had an average of 46.7 percent ground cover compared to 85.2 percent ground cover in un-grazed (livestock inaccessible or long-term rested) locations. Basal cover of grasses averaged 5.2 percent in grazed locations compared to 12.9 percent in un-grazed locations.

Table 2. Potential and historical herbaceous production, lb/acre.

Vegetation Type	Favorable Yr	Average Yr	Low Yr
Mountain stony loam aspen (USDA 1992a)	2900 – 3000 45/30 ^a	2150 – 2250 45/30	1400 – 1500 45/30
CNF Aspen (USDA 1997) ^b	--	1908/962/437 20/48	--
Mountain loam mountain big sage (USDA 1992b)	1800 – 2200 80/5	1600-2000 80/5	1200 – 1500 80/5
Mountain shallow loam mountain big sage (USDA 1992c)	1600 – 1700 50/5	1000 – 1100 50/5	500 – 600 50/5
CNF Mountain big sage (USDA 1997) ^b	--	1758/872/295 15/37	--
CNF Conifer (USDA 1997) ^b	--	1182/527/162 19/47	--
WCNF Aspen ^c	--	--	241/443/684
WCNF Mountain big sage ^c	--	--	122/163/285
WCNF Conifer ^c	--	--	157/253/410

^aPercent production by grasses/forbs.

^bCNF data are from 1959 – 1976 period and are assumed to represent the long-term average. Data are maximum/average/minimum production, including grasses, forbs and shrubs. Percent production by grasses and forbs are the average across all sites.

^cValues for the WCNF are from range analysis data sheets for the North Rich allotment for 1961 and 1962 and are in order: grasses/forbs/total herbaceous production. These data are from a below average precipitation year.

Additional locations were surveyed in the Utah portion of the Bear River Range in 2001, 2004 and 2005. These were principally in the North Rich allotment. They included three long-term un-grazed sites and 10 grazed sites in mountain big sagebrush; six grazed sites in mixed-conifer, three of which had been logged decades earlier and as a result had open canopy, and three with high canopy cover; and six grazed sites in aspen. Results of ground cover determinations at these locations are provided in table 3, while reference values are found in table 1. Grazed mountain big sagebrush locations had average ground cover of 61.8 percent compared to 94.4 percent in the un-grazed sites and 96 percent in reference sites. Grass basal cover in grazed locations averaged 3.6 percent compared to 38.9 percent in un-grazed locations. Six grazed aspen sites had 59.6 percent average ground cover compared to 98 percent for reference sites. Three mixed conifer sites that had been logged and continued to be grazed had average ground cover of 61.1 percent while three grazed closed-canopy mixed conifer sites had average ground cover of 92.2 percent. The only data

available for comparison in mixed-conifer was the CNF maximum ground cover average of 98 percent in coniferous timber. The values for all grazed sites were much lower than those for either the un-grazed sites or the reference values in table 1

Vegetation Production Surveys

Three surveys have been conducted to determine production of herbaceous vegetation in the North Rich allotment. In 2001, the survey included measurement of ground cover and plot clippings to determine production of herbaceous vegetation in mountain big sagebrush and open canopy mixed-conifer areas. In 2004, ground cover and production was assessed in additional aspen, mountain big sagebrush and high canopy mixed-conifer locations. During the period 2005 to 2007, utilization cages were installed in additional aspen, mountain big sagebrush and mixed-conifer locations to assess utilization. Caged plots were located in sites representative of average grass cover and clipped to determine production.

Table 3. Results of 2001 to 2005 ground cover surveys, percent.

Vegetation Type/Forest/Yr	Rock	Crust	Litter	Grass	Forbs	Bare Ground	Total Ground Cover
Ungrazed Reference Areas							
Mtn big sage (CNF 2001 n=3) ^a	1.4	3.5	63.6	12.9	3.9	14.8	85.2
Mtn big sage (WCNF 2001 n=3)	2.6	0.3	41.8	38.9	10.9	5.6	94.4
Grazed Areas							
Mtn big sage (CNF 2001 n=41)	2.0	0.1	34.6	5.2	5.0	53.3	46.7
Mtn big sage (WCNF 2001, 2004, 2005 n=10)	0.9	0	53.7	3.6	3.6	38.2	61.8
Aspen (WCNF 2004, 2005 n=6)	2.5	0.3	70.7	1.7	2.3	40.4	59.6
Conifer (2001 n=3) ^b	1.1	0	42.5	7.7	9.8	38.9	61.1
Conifer (2004 n=3) ^c	1.0	0.1	89.6	0.6	0.9	7.8	92.2

^a n = number of transect locations.^b Conifer area logged and thinned in prior years, low canopy cover.^c Conifer with no recent thinning, high canopy cover.**Table 4.** Grass and forb production (lb/acre) in the North Rich Allotment compared to potentials.

Vegetation Type and Year	Ppt.	Grass			Forb		
		Potential	Measured	Percent of Potential	Potential	Measured	Percent of Potential
Ungrazed Reference Areas							
Mtn big sage - 2001 (n=1) ^a	<Avg	1080	2104	195	68	94	139
Mtn big sage - 2001 (n=2) ^b	<Avg	275	432	157	28	38	138
Grazed Areas							
Mtn big sage - 2001 (n=3)	<Avg	275	118	43	28	154	560
Mtn big sage - 2004 (n=3)	Avg	525	98	19	53	159	303
Mtn big sage - 2005 (n=4)	>Avg	825	447	54	83	384	465
Mtn big sage - 2006 (n=3)	Avg	525	178	34	53	108	206
Mtn big sage - 2007 (n=2)	Avg	525	210	40	53	89	170
Aspen - 2004 (n=3)	Avg	990	140	14	660	--	--
Aspen - 2005 (n=3)	>Avg	1328	536	40	885	291	33
Aspen - 2007 (n=1)	Avg	990	160	16	660	96	15
Conifer - 2001 (n=3) ^c	<Avg	--	107	--	--	204	--
Conifer - 2004 (n=3) ^d	Avg	224	14	6	556	101	18
Conifer - 2006 (n=2) ^d	Avg	224	6	3	556	76	14
Conifer - 2007 (n=1) ^d	Avg	224	0	0	556	4	1

^a Mountain loam site.^b Mountain shallow loam sites.^c Open canopy mixed-conifer.^d Closed canopy mixed-conifer.

Precipitation records for climate stations in or adjacent to the Bear River Range were reviewed to find a station with complete data for the period of interest. Based on this review, the Richmond, Utah, station provided the most complete record, indicating that 2001 was a below average precipitation year, while 2005 was above average (WRCC 2010). The other years were near average, being slightly above

or below the long-term average. Comparisons of measured production to potential were based on this determination.

Current herbaceous production in grazed areas (table 4) was compared to potential and historical Forest Service values (table 2). Grass production measured in aspen communities during the 2000's in the North

Rich allotment ranged from 140 to 160 lb/acre during average precipitation years compared to a potential of 990 lb/acre. Forest Service range analysis data collected in the 1960's found an average of 241 lb/acre (WCNF 1962). Forb production in aspen communities was measured at only one site during an average year, finding 96 lb/acre compared to a potential of 660 lb/acre and the 1960's Forest Service data of 443 lb/acre. Mountain shallow loam big sagebrush communities produced 98 to 210 lb/acre of grass during average years compared to a potential of 525 lb/acre and the 1960's Forest Service amount of 122 lb/acre. Forb production was 89 to 159 lb/acre compared to potential of 53 lb/acre and the historical amount of 163 lb/acre during the 1960's Forest Service range analysis surveys. Mixed-conifer communities produced 0 to 14 lb/acre of grasses per year compared to the 1960's Forest Service amount of 157 lb/acre, while forbs were measured at 4 to 101 lb/acre compared to the historical amount of 253 lb/acre. If the maximum values found in the CNF range analysis for conifer were used as potentials, current production in the North Rich allotment mixed-

conifer would be well below those values. It should be emphasized that the 1960's Forest Service data from the North Rich allotment was collected during below average precipitation years, yet in most cases exceeded what is found today during average precipitation years, indicating that a decline in production may have occurred since the 1960's.

The only un-grazed, or long-term rested sites surveyed for herbaceous production were in mountain big sagebrush vegetation types (table 5). The un-grazed mountain loam site produced a total of 2,198 lb/acre total herbaceous vegetation in 2001, a below average year, compared to potential of 1,148 lb/acre. The un-grazed mountain shallow loam sites produced 470 lb/acre during a below average year compared to potential of 303 lb/acre. Grazed sites in mountain shallow loam produced 272 lb/acre in 2001, a below average year, with a range of 257 to 299 lb/acre during average years, compared to potential of 578 lb/acre. No data were collected in grazed mountain loam mountain big sagebrush areas.

Table 5. Herbaceous production surplus or deficit (lb/acre) compared to potential.

Vegetation Type and Year	Ppt.	Total Herbaceous Production		Surplus or Deficit lb/acre
		Potential	Measured	
Ungrazed Reference Areas				
Mtn big sage - 2001 (n=1) ^a	<Avg	1148	2198	1051
Mtn big sage - 2001 (n=2) ^b	<Avg	303	470	168
Grazed Areas				
Mtn big sage - 2001 (n=3)	<Avg	303	272	-31
Mtn big sage - 2004 (n=3)	Avg	578	257	-321
Mtn big sage - 2005 (n=4)	>Avg	908	831	-77
Mtn big sage - 2006 (n=3)	Avg	578	286	-292
Mtn big sage - 2007 (n=2)	Avg	578	299	-279
Aspen - 2004 (n=3)	Avg	1650	--	--
Aspen - 2005 (n=3)	>Avg	2213	827	-1386
Aspen - 2007 (n=1)	Avg	1650	256	-1394
Conifer - 2001 (n=3) ^c	<Avg	--	311	--
Conifer - 2004 (n=3) ^d	Avg	780	115	-665
Conifer - 2006 (n=2) ^d	Avg	780	82	-698
Conifer - 2007 (n=1) ^d	Avg	780	4	-776

^aMountain loam site.

^bMountain shallow loam sites.

^cOpen canopy mixed-conifer.

^dClosed canopy mixed-conifer.

Table 6. Soil organic matter, carbon, nitrogen and nitrate-N.

Year	Vegetation Type	Organic Matter percent	Organic Carbon percent	Total Nitrogen percent	Nitrate-N ppm
1992	Mtn big sage – grazed (n=7)	9.9/71 ^a	--	--	--
1992	Mtn big sage – ungrazed (n=3)	14.0	--	--	--
1995	Mtn big sage – grazed (n=5)	12.5/69	--	0.3/60	6.4/56
1995	Mtn big sage – ungrazed (n=2)	18.0	--	0.5	11.4
2001	Conifer – heavy grazing (n=2)	--	2.85/50	0.12/46	1.35/56
2001	Conifer – moderate grazing (n=2)	--	4.25/75	0.21/81	1.7/71
2001	Conifer – ungrazed (n=2)	--	5.65	0.26	2.4

^aThis value is 71% of the ungrazed value, similar for each /value.

Soil Chemistry Surveys

Soil samples were taken in 1992 and 1995 in mountain big sagebrush type, and in 2001 in mixed-conifer (table 6). In 1992, only soil organic matter (OM) was determined, with the un-grazed reference sites containing 14 percent OM and the grazed sites containing 9.9 percent OM. In 1995, sampling found 18 percent OM and 0.5 percent total nitrogen (N) in un-grazed reference sites compared to 12.5 percent OM and 0.3 percent N in the grazed sites. The mixed-conifer sites showed similar patterns of reduced soil organic matter, total nitrogen and nitrate as well as reductions in litter in grazed sites when compared to un-grazed sites. The heavily grazed site was nearest the water source (500 ft), with the moderately grazed site more distant from water (2000 ft), and the un-grazed control was in an area not accessed by livestock approximately 10,000 ft from the water source. The un-grazed site averaged 5.65 percent organic carbon compared to 4.25 percent in the moderately grazed site and 2.85 percent in the heavily grazed site. Soil total nitrogen ranged from 0.26 percent in the un-grazed site to 0.21 percent in the moderately grazed site and 0.12 percent in the heavily grazed site. Nitrate-nitrogen averaged 2.4 ppm at the un-grazed site, 1.7 ppm at the moderately grazed site and 1.35 ppm at the heavily grazed site. Litter depth averaged 2 inches in the un-grazed site, 0.8 inches in the moderately grazed site and 0.5 inches in the heavily grazed site. Both grazed sites had areas of bare soil, while ground cover was 100 percent at the un-grazed site. Only the un-grazed site had a mycorrhizal layer at the litter/soil interface.

DISCUSSION

Forest Service reference data and NRCS ecological site descriptions provided a basis for comparison to current ground cover and herbaceous vegetation

production in the Bear River Range. Ground cover potential values were obtained from the Wasatch Cache National Forest Rangeland Health EIS (USDA 1996), which presented ranges of ground cover for various plant communities including mountain big sagebrush and aspen. Data collected at sites that have been rested from grazing for long periods provided additional information for ground cover at or near potential. Maximum ground cover data from the Caribou National Forest's "Hierarchical Stratification of Ecosystems for the Caribou National Forest" (USDA 1997) sites were also considered as potentials. These closely aligned with the upper limits of reference published by the WCNF (USDA 1996). These were provided in table 1.

Grazed areas surveyed in mountain big sagebrush, aspen and previously thinned mixed-conifer forest had ground cover ranging from 46.7 percent to 61.8 percent, compared to potential values of greater than 90 percent. Only high canopy mixed-conifer forest, at 92.2 percent ground cover, approached potential. This was likely due to the absence of sufficient forage to attract livestock and thereby reduced the presence of livestock and associated grazing and trampling, which allowed litter to accumulate and cover the soil. When current ground cover was compared to historical Forest Service values from the 1960's, conditions did not appear to be improved and may have declined (table 1; table 3). When measurements were taken with increasing distance from water, ground cover increased, indicating that reduced grazing intensity was correlated with increased ground cover (figure 2).

These reduced levels of ground cover lead to increased soil erosion as predicted by the literature. Analysis of two allotments in the Bear River Range in Idaho used tree and shrub canopy measurements, ground cover data, precipitation values and the

Universal Soil Loss Equation (Ruhe 1975) to determine relative erosion rates at different ground cover values (Carter et al. 2006). The analysis determined that the reduced levels of ground cover in the Bear River Range result in high rates of erosion. At the Caribou National Forest ground cover standard of 60 percent, erosion was up to 15 times higher than background. The levels of ground cover found in this study were near this level and would result in similar high levels of erosion (table 3).

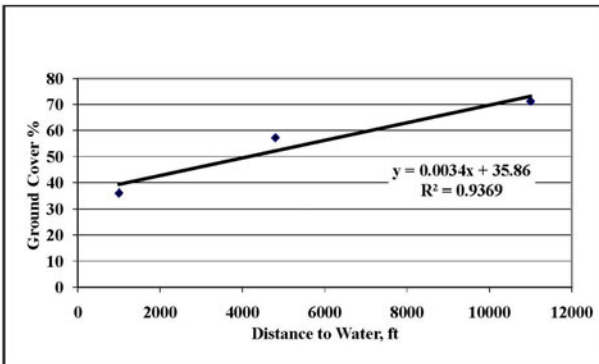


Figure 2. Ground cover vs. distance to water (2004 data).

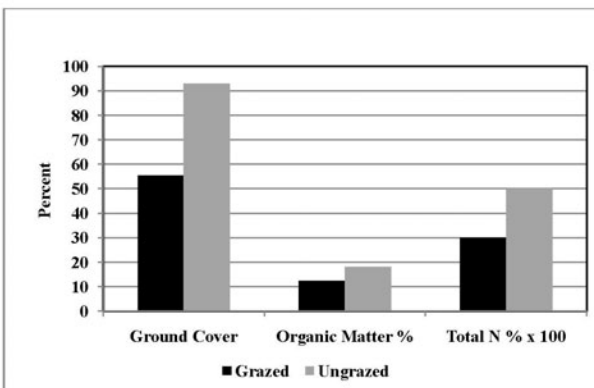


Figure 3. Ground Cover vs. Soil Organic Matter and Total Nitrogen (1995 data).

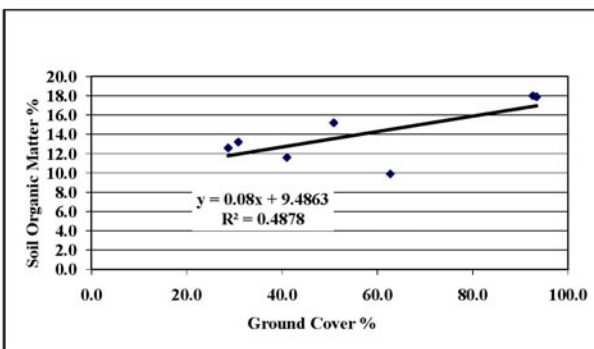


Figure 4. Soil organic matter vs. ground cover (1995 data).

This accelerated erosion carries the nitrogen and carbon contained in surface soils down-gradient, thereby reducing the pool of carbon and nitrogen stored in the forest. Soil samples taken in un-grazed and grazed mountain big sagebrush locations in the Bear River Range in 1992 and 1995 showed that organic matter was reduced by approximately 30 percent, total nitrogen by 40 percent and nitrate-N by 44 percent in grazed areas compared to un-grazed areas (table 6; figure 3). When the 1995 data for soil organic matter and nitrogen were plotted against ground cover, a positive correlation was found, indicating higher ground cover was associated with higher soil organic matter and total nitrogen (figures 4 and 5). A similar pattern of decline of soil organic carbon, total nitrogen, nitrate-nitrogen and litter depth occurred in samples taken from grazed sites in mixed-conifer forest compared to an un-grazed site (table 6; figure 6). The heavily grazed site, when compared to the un-grazed site, showed a decline in organic carbon of 50 percent, total nitrogen by 54 percent, and nitrate-N by 44 percent. Litter depth in the heavily grazed site was 25 percent of that in the un-grazed site and only the un-grazed site had an evident and complete mycorrhizal layer at the litter and soil interface.

Production measurements and comparisons to potential were provided in Table 4. Grass production in un-grazed mountain big sagebrush sites ranged from 157 to 195 percent of potential, while forbs were at 138 to 139 percent of potential. In grazed mountain big sagebrush sites, grass production ranged from 19 to 54 percent of potential, while forbs ranged from 170 – 560 percent of potential, reflecting dominance by non-palatable species, or increasers, which are avoided by livestock. Grasses in grazed aspen sites ranged from 14 to 40 percent of potential production, while forbs ranged from 15 to 33 percent of potential. If the CNF historical maximums were used for comparison, mixed-conifer grass production ranged from 0 to 14 percent of potential and forbs ranged from 1 to 18 percent of potential. When the 2004 grass production data was plotted against ground cover, a positive correlation was found, indicating that grass production increased as distance from water increased. This reflected the reduced intensity of grazing further from the water source (figure 7).

The surplus or deficit of total herbaceous production compared to potential was provided in table 5. The un-grazed site in mountain loam mountain big sagebrush produced a surplus of 1,051 lb/acre and a surplus of 168 lb/acre in the shallow loam sites. No grazed sites produced a surplus compared to potential. The deficit in grazed mountain big sagebrush communities ranged from 77 to 321 lb/acre. The deficit in aspen communities ranged from 1,386 to 1,394 lb/acre. Mixed-conifer, when compared to the maximum values found in the CNF had deficits ranging from 665 to 776 lb/acre. These figures don't take into account the belowground portion of plants.

Holechek and others (2004) reported that total nitrogen in Australian livestock forage ranged from 1.4 to 2.2 percent. Haferkamp and others (2005) found nitrogen concentrations in mixed grass prairie varied through the seasons, ranging from 1.7 percent in spring to 0.75 percent in fall for mixed grasses and forbs. Qi and others (2008) found that grasses and forbs in lightly degraded areas had carbon content of 42.0 and 42.5 percent and nitrogen content of 1.34 and 1.41 percent, while in heavily degraded areas, carbon declined to 37.3 and 40.5 percent with nitrogen values of 1.31 and 1.38 percent respectively. Based on these literature values for carbon and nitrogen in livestock forage, values of 43 percent carbon and 1.4 percent nitrogen contained in herbaceous plants were used to estimate the potential pool of carbon and nitrogen present in the above-ground portion of herbaceous vegetation sampled. According to West (1983) root masses can constitute up to half the biomass present in sagebrush vegetation types.

The values for carbon and nitrogen content in herbaceous vegetation were applied to the literature values for potential production of herbaceous vegetation in the plant communities found in the Bear River Range to estimate potential storage. Based on this, significant potential for carbon and nitrogen storage exists within the plant communities (table 7). Calculated carbon and nitrogen values based on potential herbaceous production for each vegetation type were compared to long-term un-grazed sites and grazed sites. Long-term un-grazed sites were in a surplus for both carbon and nitrogen while grazed sites were in a deficit. The surplus in mountain big

sagebrush un-grazed sites ranged from 72 to 451 lb C/acre and 2.3 to 14.7 lb N/acre. The deficit for mountain big sagebrush sites in grazed areas ranged from 13 to 138 lb C/acre and 0.4 to 4.5 lb N/acre. The deficit in grazed aspen ranged from 596 to 600 lb C/acre and 19.4 to 19.5 lb N/acre. The deficit for mixed-conifer (based on CNF maximum production values) ranged from 286 to 333 lb C/acre and 9.3 to 10.8 lb N/acre.

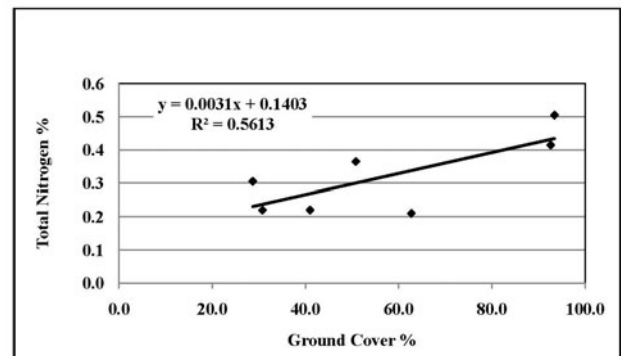


Figure 5. Soil total nitrogen vs. ground cover (1995 data).

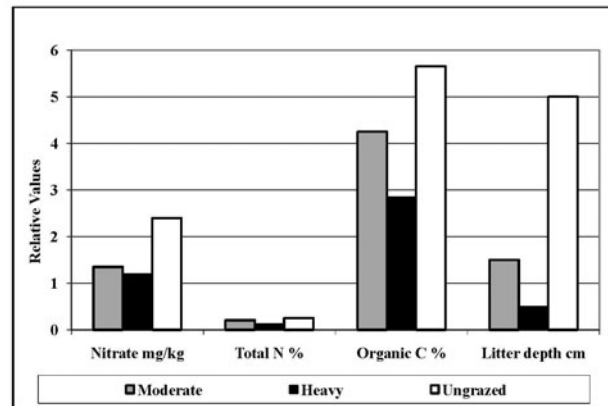


Figure 6. Soil conditions vs. grazing intensity.

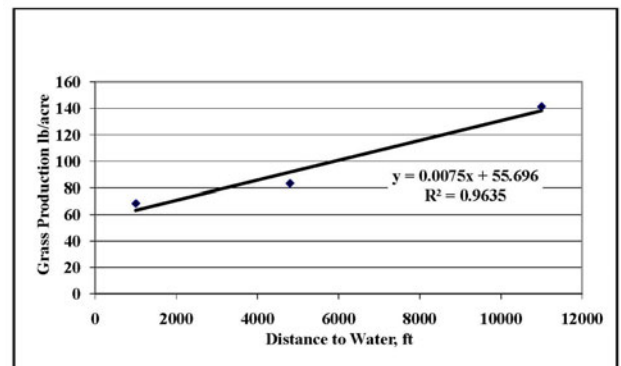


Figure 7. Production of grasses vs. distance to water (2004 data).

Table 7. Surplus or deficit of organic carbon and nitrogen compared to potential.

Vegetation Type	Ppt.	Potential Organic C lb/acre	Estimated Organic C lb/acre	Surplus or Deficit OC lb/acre	Potential Total N lb/acre	Estimated Total N lb/acre	Surplus or Deficit N lb/acre
Ungrazed Reference Areas							
Mtn big sage - 2001 (n=1) ^a	<Avg	494	945	451	16.1	30.8	14.7
Mtn big sage - 2001 (n=2) ^b	<Avg	130	202	72	4.2	6.6	2.3
Grazed Areas							
Mtn big sage - 2001 (n=3)	<Avg	130	117	-13	4.2	3.8	-0.4
Mtn big sage - 2004 (n=3)	Avg	249	111	-138	8.1	3.6	-4.5
Mtn big sage - 2005 (n=4)	>Avg	390	357	-43	12.7	11.6	-1.1
Mtn big sage - 2006 (n=3)	Avg	249	123	-126	8.1	4.0	-4.1
Mtn big sage - 2007 (n=2)	Avg	249	129	-120	8.1	4.2	-3.9
Aspen - 2004 (n=3)	Avg	710	--	--	23.1	--	--
Aspen - 2005 (n=3)	>Avg	952	356	-596	31.0	11.6	-19.4
Aspen - 2007 (n=1)	Avg	710	110	-600	23.1	3.6	-19.5
Conifer - 2001 (n=3) ^c	<Avg	--	134	--	--	4.4	--
Conifer - 2004 (n=3) ^d	Avg	335	49	-286	10.9	1.6	-9.3
Conifer - 2006 (n=2) ^d	Avg	335	35	-300	10.9	1.1	-9.8
Conifer - 2007 (n=1) ^d	Avg	335	2	-333	10.9	0.1	-10.8

^aMountain loam site.^bMountain shallow loam sites.^cOpen canopy mixed-conifer.^dClosed canopy mixed-conifer.

These data show that in areas of the Bear River Range surveyed, ground cover, herbaceous production, carbon and nitrogen storage have been reduced below potential and likely continue to decline, whereas areas rested from livestock grazing for long periods have ground cover and production at or near potential and contain a significant reservoir of stored carbon and nitrogen. Rest from grazing has not been provided in the study area, yet is essential to recover degraded plant and soil communities. For example, native bunchgrass species, such as bluebunch wheatgrass (*Pseudoroegneria spicata*) and Idaho fescue (*Festuca idahoensis*), which are key species in sagebrush-steppe ecosystems, require several years of rest following each period of grazing in order to restore their vigor and productivity (Anderson 1991; Clary and Webster 1989; Hormay and Talbot 1961; Mueggler 1975). The recovery of degraded plant and soil communities can take many years, even under total rest (Anderson and Inouye 2001; Orr 1975; Owens and others 1996; Trimble and Mendel 1995).

Grazing management in the study area has relied upon installation of water developments and grazing systems. For example, the North Rich allotment contains over 130 water developments yet ground cover, herbaceous production, soil carbon and

nitrogen are well below potential. Stocking rates have not been adjusted to reflect current forage availability and forage consumption rates, yet research has shown that it is reductions in stocking rate that lead to increased production, not grazing systems (Briske and others 2008; Clary and Webster 1989; Holechek and others 1999; Van Poolen and Lacey 1979). Utilization rates commonly used by the Forest Service and other agencies have remained near 50 percent in spite of research that shows utilization levels in the range of 25-30 percent should be used to maintain productivity (Galt and others 2000; Holechek and others 2004).

The Forest Service has not conducted forage capacity surveys since the early 1960's. Galt and others (2000) recommended that grazing capacity surveys should take place at intervals of no more than 10 years and that grazing capacity determinations take into account slope (<30 percent) and distance to water (< 1 mile) limitations. Forage consumption rates currently used by the Forest Service and other agencies underestimate the demand from today's larger cattle by using 26 lb/day, or 780 lb/month forage consumption for an animal unit month (AUM), which is considered to be one cow/calf pair or five sheep with lambs. Today's cow/calf pair weighs

approximately 1,680 pounds, while a ewe/lamb pair weighs 275 pounds (Carter 2008). Cattle consume 3 percent of their body weight in air-dry forage per day (USDA 2003c), while sheep consume 3.3 percent (USDA 1965). Applying these rates to the combined weight of the cow/calf pair gives a forage consumption rate of over 50 lb/day or 1,532 lb/month air-dry forage. A similar analysis for sheep leads to a consumption rate of 9.1 lb/day for each ewe/lamb pair, which for five ewe/lamb pairs is 1,380 lb/month air-dry forage. These values are nearly twice those used by the land management agencies for an AUM.

CONCLUSIONS

The analysis presented here illustrates the current degraded state of plant communities and soils in grazing allotments in the Bear River Range. The lack of science-based livestock grazing management has resulted in the loss of native grass and forb production, shifts to less palatable and more grazing-tolerant species, and large decreases in ground cover from potential. The consequence has been increased soil erosion and the loss of carbon and nitrogen storage in soils as well as in the herbaceous components of plant communities. This observed loss in native plant productivity as a result of livestock grazing practices is not unique to the Bear River Range (Catlin and others in press).

Implementing restoration practices and science-based grazing management on the 305,000,000 acres of public lands and 220,000,000 acres of Indian Reservations and private lands grazed by livestock in the eleven western states has the potential to restore native plant communities and store significant amounts of carbon and nitrogen to mitigate the impacts of climate change. Other benefits would include improved watershed function, enhanced water supplies, lowered water treatment costs, and healthy fish and wildlife populations. The costs of continued livestock grazing should be evaluated against the value of these and other restored ecosystem services. Reliance on failed livestock grazing strategies must be reversed and mechanisms must be found to provide for long-term rest sufficient to recover these degraded systems to potential. This can be accomplished through allotment and/or pasture closures through voluntary action, mandate, or by permit buyouts. Education of livestock producers and providing incentives for carbon storage on private lands and Indian Reservations, much like the

Conservation Reserve Program or Grassland Reserve Program managed by NRCS may have the potential to offset some of the losses from those lands.

Where livestock grazing continues on public lands, the series of steps below must be taken to ensure that it is sustainable and the plant and soil communities are restored to potential with their associated potential ground cover, production of native species, carbon and nitrogen storage.

Determine available grazing capacity based on surveys of current forage production by desirable herbaceous species and factors such as slope (<30 percent) and distance to water (<1 mile) with areas of sensitive or high erosion hazard soils being eliminated from stocking rate calculations.

Update stocking rates based on conservative utilization rates of 25 to 30 percent and current forage consumption rates of cattle and sheep.

Manage all livestock by herding instead of relying on additional pasture fencing and water developments, which have not succeeded and have resulted in increased range degradation where these have been installed.

Provide adequate rest for plants to recover vigor and productivity after being grazed and before being grazed again. This can require several years of rest for each grazing period. Productivity should be monitored prior to grazing to ensure recovery.

Provide long-term rest for recovery of degraded soil and plant community productivity. Where multiple pastures already exist, single pastures could be excluded from grazing until restored, then grazed again under the preceding principles. When areas are closed, stocking rates must be adjusted downward based on the remaining capacity of the allotment.

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Range Management in the Face of Climate Change

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ABSTRACT

Climate change forecasts predict more frequent and more intense droughts in the West. These droughts will significantly impact wildlife habitat. Today most of our western rangelands are impaired. If restored, the predicted impacts of drought, and thereby, climate change, could be significantly reduced on our rangelands. This study evaluates how the Department of the Interior is measuring ecological health on rangelands and whether agency management effectively restores habitat's resilience, or ecological potential. This in-depth case study of a Bureau of Land Management (BLM) allotment in Utah reviews agency methods and uses five years of the authors' field data to understand if and how current BLM range management is addressing impacts to habitat from climate change. BLM does not inventory the ecological health and resilience of rangelands, and its qualitative ecological assessment methods are inadequate to identify or measure key ecological conditions. While we, as a society, have the capability to manage livestock grazing to restore habitat, the results of our case study shows this is not happening fast enough on the scale needed and degraded habitat is often under reported. Where agency management identifies problems, agency responses often rely on internal faulty habitat information. We found that fewer livestock actually grazed the allotment than were reported, BLM underestimated utilization, and also failed to adequately monitor trend and upland and riparian health. Our capacity analysis, based on forage production, cattle weights and sustainable utilization, determined that the number of livestock permitted is six times more than the carrying capacity of the study allotment. Habitat restoration must be part of the response to climate change. To achieve this, significant changes in range management on western rangelands will be needed.

In Monaco, T.A. et al. comps. 2011. *Proceedings – Threats to Shrubland Ecosystem Integrity*; 2010 May 18-20; Logan, UT. Natural Resources and Environmental Issues, Volume XVII. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan Utah, USA.

INTRODUCTION

Climate change is likely to lead to longer and more intense droughts in the Southwestern U. S. (IPCC 2007). The combination of climate change and habitat impairment represents one of the most potentially serious problems that humans, wildlife and their habitat have ever faced (Root et al. 2003). Severe impacts to ecosystem services are predicted, exacerbating the impacts from current natural and human stress factors (Bate and others 2009).

To date, the responses to climate change have focused primarily on mitigating climate-influencing gas emissions caused by human activities (Climate Action Network 2009). However, the use of range management to control the adverse effects of climate change has been largely neglected. What role does range management have in responding to climate change?

Actions that reduce the vulnerability of natural systems to climate changing influences have been

recommended as a means of coping with climate change (IPCC 2007). These actions can include creating redundant populations, maximizing core areas and connectivity, and increasing habitat resilience (Malcolm and Pitelka 2000, Running and Mills 2009). C.S. (Buzz) Holling introduced the concept of resilience in ecological systems, defining resilience as a measure of how far the system could be perturbed without shifting to a different state (Holling 1973, Gunderson and Holling 1997). Increased habitat resilience helps ecosystems better withstand climate change (Bate et al. 2009).

Rangelands play an important role in regulating atmospheric carbon. Worldwide, soil organic matter contains three times as much carbon as the atmosphere (Ecological Society of America 2000, Allmaras et al. 2000, Flynn et al. 2009). Long term intensive agriculture can significantly deplete soil organic carbon (Benbi and Brar 2009). Past rangeland use in the United States has led to similar losses (Follett and others 2001, Neely and others 2009). Soil organic carbon is an important source of

energy that drives many nutrient cycles. Increases in soil organic carbon and other organic matter lead to greater pore spaces and more soil particle surface area which retains more water and nutrients (Tisdale and others 1985). Soil organic carbon, which makes up about 50 percent of soil organic matter, is correlated with soil fertility, stability, and productivity (Herrick and Wander 1998).

The future impacts of climate change on western rangelands are predicted to be driven by more severe droughts (IPCC 2007). According to the U.S. Drought Monitor, which assesses the severity of droughts based on precipitation and soil moisture (Palmer 1965, Wilhite 2005), habitat impacts and vulnerability increase with drought intensity (Wilhite and others 2007). According to the National Drought Mitigation Center (2010), a moderate drought (D1) will cause some damage to plants, a high fire risk, and water shortages. An extreme drought (D3) leads to major plant loss, extreme fire danger, and likely widespread water use restrictions.

Models used to predict changes in species' ranges due to climate change often describe changes in environmental conditions of habitat based on changes in parameters that drive those environmental conditions (Pearson and others 2006). Today, a majority of western rangelands are in degraded condition and thus the predicted impacts of climate are also based on habitat that has been degraded. As a result, a common unstated assumption of the nine models that Pearson and others (2006) tested is that habitat resilience will be the same in the future as it is today. Clearly, modeling is needed that is based on habitat that is not degraded. We would predict that such modeling (of lands at their ecological potential) will show far fewer impacts than for impacted lands.

There has been little research that compares the impacts of drought on habitat that has lost its resilience with similar habitat that has not (Peterson 2009). Two examples from the Escalante River basin, Utah, offer some insight into the connection between drought and habitat resilience. The Gulch, a perennial stream in the Grand Staircase Escalante National Monument (figure 1) has almost no shading, is shallow and wide with mostly bare banks, resulting in high summer water temperatures. Fish and amphibians are absent. Five miles away is another perennial stream, Deer Creek (figure 2). The cross section of this stream channel resembles the bottom

of an hour glass, narrow at the top and wide at the bottom. Mostly shaded, this stream supports persistent populations of both fish and frogs. Both streams are similar in many ways. The geology, soils, elevation, and climate are similar for both sites; thus, they should possess similar habitat characteristics. However, livestock grazing in Deer Creek has rarely occurred for the past 50 years, whereas 300 cow/calf pairs graze in The Gulch from November through March of each year (BLM 2008c). Deer Creek is near its ecological potential, and has resilience. The photos in figures 1 and 2 were taken during a D1 severity drought that has lasted most of the past seven years.



Figure 1. The Gulch (stream) during a drought in 2007. Photo BLM.



Figure 2. Deer Creek during a drought in 2007. Photo David Smuin.

Places like Deer Creek are rare. Most of the streams in the Intermountain West are in a degraded condition similar to that found in The Gulch (Belsky and others 1999, Baker and others 2003, BLM 2005, Milchunas 2006). Most rangelands in the West have been

significantly impacted by human activities in the past and remain impaired today (Cottam 1945, PRIA 1978, Burkhardt 1996, BLM 2002, Baker and others 2003, Milchunas 2006). Riparian areas are often impacted by traditionally practiced livestock grazing (Platts 1991, Ohmart 1996) leaving approximately 80 percent of streams and riparian areas damaged in the western United States (Belsky 1999). According to the American Fisheries Society, 15,000 of 19,000 miles, or 77 percent of streams on BLM land are in unsatisfactory condition (Armour and others 1994). The Forest Service states that "Riparian areas throughout the Intermountain Region have been significantly affected over the past several decades. Most of these effects have been negative, including: lowering of water tables, erosion of stream channels, exotic plant encroachment (e.g. tamarisk), removal of beaver populations, concentrated runoff and increased sediment from road construction, and changes in vegetation composition" (Forest Service 1996).

The second example involves Twin Creeks and Mill Hollow, two similar sagebrush steppe habitats in the Uinta-Wasatch-Cache National Forest. In 2007, during a D2 intensity drought, site productivity was measured using the paired plot method (BLM 1996a) at both sites. Grass samples taken at Twin Creeks averaged 1023 kg/hectare air dry weight. This is similar to grass production expected during an unfavorable year for a site in excellent condition or at its ecological potential (Mason 1971). Mill Hollow had grass production of 139 kg/hectare air dry weight or 13 percent of that found at the Twin Creeks site. Soil, elevation, and climate conditions at these two sites are similar. Livestock grazing in Twin Creeks involves trailing for just five days a year, while Mill Hollow is grazed by 300 cow/calf pairs from late June to mid September annually (USFS 2004). This example shows that even during a drought, a site near its ecological potential shows a high level of herbaceous plant productivity, significantly more than that of habitat under typical grazing management.

These two examples demonstrate the hypothesis that habitat near its ecological potential is less vulnerable to climate change than habitat below its ecological potential (Beschta 1987). Thus, the restoration of habitat resilience becomes an important response to climate change. The field of restoration ecology has recently made significant advances in developing the needed theory for restoration (Falk and others 2006);

and with better data on the ecological condition of habitat, we can better describe what is needed to achieve recovery of degraded sites. With a new focus on identifying habitat that has lost its resilience, followed by actions for restoration, we can reduce the severity of the impacts from the intense droughts that are forecast for the West.

Climate Change: BLM Ecological Assessments For Meeting Rangeland Health Standards

The Department of the Interior has taken steps to integrate climate change into its programs. The Secretary of the Interior signed Secretarial Order 3226 (DOI 2009), requiring Interior bureaus to analyze climate change in plans and policies. In 2007, Secretary Kempthorne initiated a Climate Change Task Force to report on climate change impacts and strategies relevant to Department of Interior lands. The need to restore habitat resilience was not included among the adaptation opportunities described in this report (Neely and Wong 2009, USGS 2008). In September of 2009, Secretary Salazar issued Secretarial Order 3289 that revised the direction that the Department of the Interior would take in addressing climate change (Salazar 2009a). This order called for coordination among federal agencies to promote three functions – renewable energy production, carbon capture and storage, and climate adaptation (Salazar 2009b). This order established the Climate Change Response Council and eight Climate Change Response Centers to develop response strategies that federal agency Landscape Conservation Cooperatives would act upon. So far, the new Council and Centers have not used the term "resilient habitat," or discussed the need to restore habitat as a part of adaptation or carbon storage strategies (Haynes 2009). BLM's 2008 science strategy does not mention climate change as part of the agency's priorities (BLM 2008a). However, BLM's 2010 budget does include funding for agency response to climate change (BLM 2009a).

To respond to climate change, it makes sense to review the relevance of past ecological assessment methods that BLM uses in the context of habitat resilience. For more than a decade, the BLM has had ecosystem management policies in place. Rangeland Reform '94 established national standards for range management to address ecosystem health (BLM 204a, DOI 2004; Nicoll 2005). Each state BLM office has established Rangeland Health Standards, based

on these national standards, designed to maintain functioning ecosystems. Utah's rangeland health standards open with, "It is time for change, and BLM is changing to meet the challenge. BLM is now giving management priority to maintain functioning ecosystems. This simply means that the needs of the land and its living and nonliving components (soil, air, water, flora, and fauna) are to be considered first" (BLM 1997). These Standards require that managers make significant progress in four areas: watersheds are in properly functioning condition, ecological processes are maintained, water quality meets state standards, and habitats are meeting special status species needs.

BLM's handbook H-4180-1 (BLM 2001b) describes the practices that BLM follows to implement the Rangeland Health Standards (43 CFR 4180). BLM first conducts an evaluation and then makes a determination of whether rangelands are in properly functioning condition (Standards are met) or functioning at risk (one or more Standards are not met). Where Standards are not met, BLM must determine whether livestock grazing is a factor. If the area is not making significant progress towards meeting Standards and livestock is a factor, change in livestock management is required no later than the next grazing year. To collect field data and assess whether rangelands are in properly functioning condition, BLM relies primarily on the field assessment methods described in three technical references, "Interpreting Indicators for Rangeland

Health" (Pellant and others 2000), "Process for Assessing Proper Function Condition for Lentic Riparian-Wetland Areas" (Prichard 2003a), and "A Guide to Assessing Proper Function Condition and the Supporting Science for Lotic Areas" (Prichard 2003b).

Each year BLM compiles the results of all rangeland health assessments (BLM 2009c) in a published report title "Rangeland Inventory and Monitoring Evaluation Report." The results for 2009 are presented in table 1. In Utah, with 68 percent of 1,413 BLM allotments evaluated, 1 percent were not meeting standards or making significant progress towards meeting standards and livestock use was a factor. This means that BLM argues that only a very small number of allotments, 1 percent of the assessed Utah BLM allotments, require changes in grazing management in order to meet rangeland health standards.

Responding to climate change requires assessing the condition of habitat and then responding to stressors. To assess the impact of range use, BLM conducts range monitoring, including trend, utilization, and ecological site inventory, which supports annual grazing management decisions. Permanent trend sites, where data are gathered periodically, are established in most allotments.

Table 1. National assessment of BLM allotments that met the Standards for Rangeland Health as of 2009.

Category	Total BLM allotments (% of assessed)	Utah BLM allotments (% of assessed)
A. Rangelands meeting all standards or making significant progress toward meeting the standard	11,603 (78%)	813 (80%)
B. Rangelands not meeting all standards or making significant progress toward meeting the standards but appropriate action has been taken to ensure progress toward meeting the standards. Livestock is a significant factor.	1,620 (11%)	132 (13%)
C. Rangelands not meeting standard or making significant progress toward meeting the standards and no appropriate action has been taken. Livestock is a significant factor.	335 (2%)	9 (1%)
D. Rangeland not meeting all standards or making significant progress toward meeting the standards due to causes other than livestock grazing.	1,318 (9%)	65(6%)
Total number of allotments that have been assessed	14,876	1,019
Total number of allotments	21,363	1,408

Source: Bureau of Land Management. 2010. Rangeland Inventory, Monitoring, and evaluation Report, Table 7 Standards for rangeland health cumulative accomplishments.

A number of data collection methods are commonly used on these trend sites, including nested frequency data on plant species and canopy, photo plots, and line intercept transects (BLM 1996b). At the trend sites, BLM often focuses on “key species,” usually important forage plants (BLM 1984a, BLM 1989, Elzinga and others 1998).

Annual utilization monitoring relies primarily on observer estimates of the percent of key species that have been removed by livestock and wildlife. This “key species method of herbaceous removal” (BLM 1984c, 1996a) requires that the observer classify the utilization of a key species at a site based on qualitative descriptions. In riparian areas, stubble height data for key plants may be collected to assess utilization (BLM 1996a). The end-of-season reports that the grazing permit holder is required to submit are the most common record of grazing practices conducted on an allotment, which lead to the observed utilization levels.

Based on monitoring, BLM can make changes in the number of livestock to be permitted in an allotment, the season of use, and the length of grazing season (BLM 1984d, BLM 1989). Other potential changes include whether to manipulate vegetation for the benefit of livestock, and whether to construct range improvements (e.g., fences, grazing exclosures, ponds, pipeline with troughs, etc.). BLM also makes decisions on the grazing system, such as rest rotation or deferred rotational grazing.

Do the management tools used by the BLM for range management adequately assess habitat resilience and guide the required response? Because of the breadth of this topic, this paper uses a comprehensive analysis in order to answer this question. Based on the authors’ long-term study of a BLM grazing allotment in northern Utah, we are able to explore the ability of BLM’s methods to assess rangeland health.

METHODS

Study Setting, Duck Creek Allotment

The Duck Creek Allotment is located in Rich County in northeastern Utah. This area is part of the Intermountain Region, Middle Rocky Mountain Physiographic Province Wasatch Mountain Floristic Zone, which extends for over 200 miles north to south (Cronquist and others 1972). This zone is recognized

as a key wildlife corridor connecting the Greater Yellowstone Ecosystem in the north to the Uinta Mountains and southern Rockies in the south (USFS 2003). It is a semi-arid cold desert sagebrush-grassland, or sage-steppe type, in which the majority of the precipitation falls as snow during late fall to early spring, while summers are dry (Holechek and others 2004).

The Duck Creek allotment lies in the Bear River Plateau which contains nearly level to steep uplands dissected by numerous small drainages. These small streams range from perennial to ephemeral. Many are diverted or dammed into reservoirs for irrigation before reaching the Bear River. Annual precipitation varies from approximately 305 mm/year (12”) at lower elevations to 406 mm/year (16”) at higher elevations (SCS 1982). Temperatures range from a minimum monthly average of -17° C in January to a maximum monthly average of 27° C in July (Western Regional Climate Center 2010). During the 26-year period 1982 to 2009, the nearest climate station (14 km south), recorded 15 years with below average precipitation (figure 3). During the period 2000 to 2009, the U.S. Drought Monitor assessed three years as normal with seven years in various stages of drought (U.S. Drought Monitor 2010).

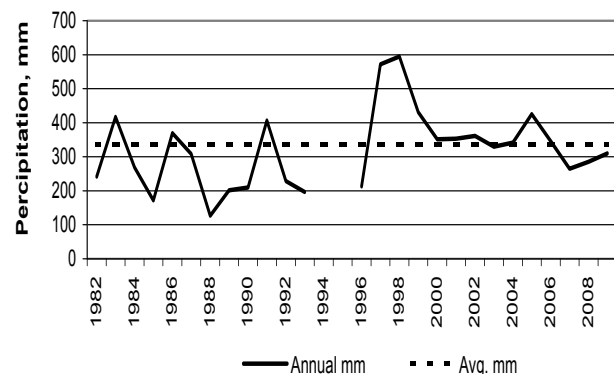


Figure 3. Annual Precipitation for Randolph, Utah, near the Duck Creek Allotment.

Elevations on the Duck Creek allotment range from 1,920 to 2,220 meters. The allotment contains 9,053 ha (22,371 acres) of which 5,297 ha are BLM lands, 3,474 ha are private, and 427 ha are State lands. Perennial streams on BLM lands within the allotment include Duck Creek, Six Mile Creek and North Fork Sage Creek. Twenty-nine springs occur on BLM lands within the allotment (BLM 2008b).

The plant community consists of shrubs dominated by sagebrush, including: Wyoming big sagebrush (*Artemisia tridentata wyomingensis*), low sagebrush (*A. arbuscula*), black sagebrush (*A. nova*), basin big sagebrush (*A. t. tridentata*), green rabbitbrush (*Chrysothamnus viscidiflorus*), bitterbrush (*Purshia tridentata*), serviceberry (*Amelanchier utahensis*), snowberry (*Symphoricarpos oreophilus*), and winterfat (*Krascheninnikovia lanata*). Small groves of aspen (*Populus tremuloides*) and Utah juniper (*Juniperus osteosperma*) are present. Willow (*Salix spp.*) are rare in riparian areas, which are dominated by Kentucky bluegrass (*Poa pratensis*), redtop (*Agrostis spp.*), and Nebraska sedge (*Carex nebraskensis*). Perennial grasses present include: bluebunch wheatgrass (*Pseudoroegneria spicata*), Indian ricegrass (*Orozopsis hymenoides*), western wheatgrass (*Pascopyrum smithii*), and Sandberg's bluegrass (*Poa Sandbergii*). Broad-leaved flowering plants include: arrowleaf balsamroot (*Balsamorhiza sagittata*), buckwheat (*Eriogonum spp.*), spiny phlox (*Phlox hoodii*), pussytoes (*Antennaria microphylla*), and yarrow (*Achillea millifolium*). Some areas on south-facing slopes are invaded by cheatgrass (*Bromus tectorum*) and noxious weeds such as black henbane (*Hyoscyamus niger*), Canada thistle (*Cirsium arvense*), musk thistle (*Carduus nutans*), and houndstongue (*Cynoglossum officinale*) occur in valley bottoms. Based on herbaria collections, the Utah Plant Atlas identified 131 vascular plant species as occurring in the Duck Creek Allotment (Ramsey and others 2004, Schultz and others 2006).

The Duck Creek allotment contains habitat for BLM sensitive species including sage grouse (*Centrocercus urophasianus*), short-eared owl (*Asio flammeus*), ferruginous hawk (*Buteo regalis*), golden eagle (*Aquila chrysaetos*), and pygmy rabbit (*Brachylagus idahoensis*) (BLM 2008b). Large ungulates include mule deer (*Odocoileus hemionus*), Rocky mountain elk (*C. canadensis nelsoni*) and pronghorn (*Antilocapra americana*). Small mammals include white-tailed jackrabbit (*Lepus townsendi*), cottontail (*Sylvilagus nuttali*), yellow bellied marmots (*Marmota flaviventer*), Uinta ground squirrels (*Citellus armatus*), least chipmunk (*Eutamias minimus*), and badger (*Taxidea taxus*). Over 90 migrant bird species that occur in the area include Brewer's sparrow (*Spizella breweri*), sage sparrow (*Amphispiza belli*), and sage thrasher (*Oreoscoptes montanus*) (BLM 1980a, b).

Eight range sites occur on the allotment: mountain loam, semidesert loam, semidesert stony loam, upland loam, upland shallow loam, upland shallow loam (juniper), upland stony loam, and woodland (aspen). The soil survey for this allotment identifies 26 different soil map units which are dominated by high or very high erosion hazard (SCS 1982). Riparian areas are not described in the soil survey, but are associated with the springs and streams. The streams have become incised and have lost access to their historical floodplains.

Livestock, including cattle, sheep and horses, have grazed Rich County and the Duck Creek allotment since settlement of the area in the 1800's. Currently six individual permits allow 400 cattle, 14 horses, and 765 sheep to graze on BLM lands and an additional 241 cattle and 305 sheep are allowed under exchange of use with private and state lands within the allotment boundary. The grazing season for cattle is May 10 thru September 7. Sheep graze under two permits, during spring from May 10 to July 1 and in fall from September 20 until December 1. Total AUMs under Active Use are 2,134 with an additional 1,176 allowed under Exchange of Use, for a total permitted use of 3,310 AUMs (BLM 2004b, 2008b).

Structural range facilities include the allotment boundary fence and two internal pasture fences that divided the allotment into four pastures in 2006. Prior to that time, the allotment lacked internal pasture fences. Water developments on BLM lands include fourteen troughs, eleven spring developments and six excavated ponds. (BLM 2009b).

Authors' Data Collection Methods

In 2001, BLM determined that the Duck Creek Allotment did not meet the Standards for Rangeland Health (BLM 2001c). In response to a long-term regional drought and issues raised by some members of the conservation community (Carter and Bloch 2001), in 2002 Rich County initiated a collaborative process to improve wildlife habitat and livestock grazing management in the county (Rich County 2007).

The Duck Creek Study area was chosen by the Rich County Coordinated Resource Management Collaboration (CRMC) as a priority area for implementing practices to achieve their goals for improved management of wildlife and ranching. To implement these goals, the CRMC developed a multi-

pasture rotation proposal with new upland water troughs and a distribution system (BLM 2004b), which BLM proposed to adopt in a Draft Allotment Management Plan for Duck Creek in 2004 (BLM 2004b). A modified proposal (BLM 2008b) was implemented in 2009, with construction of a 14 km pipeline and 6 additional watering locations in the southern half of the allotment.

The Utah Division of Wildlife Resources and others began studies focused on the Duck Creek Allotment beginning in 2005 (Norvell 2008). In 2005, the CRM established a monitoring committee. Working with this monitoring committee, the authors developed a monitoring plan that would augment other data being collected in this allotment. This study presents the data collected from 2005 to 2009 on herbaceous plant annual production and utilization, riparian residual stubble heights, canopy and ground cover, water quality, and number of cattle on the allotment.

Herbaceous Plant Annual Production and Utilization

The upland herbaceous plant community was sampled using the paired plot method (BLM 1996a). Utilization cages (1.2 m²) were placed in riparian and upland locations prior to the start of livestock grazing (figure 4). These cages excluded herbivory by rabbits and larger animals. Sampling sites were chosen to represent soil map units that covered a majority of the allotment, key range sites identified by BLM, riparian areas, and Utah Division of Wildlife Resources wildlife survey sites. At each location, a sample frame (0.84 m² or 9 ft²) was used inside the cage and on ten sites outside the cage to establish plots within which total residual herbaceous plant biomass was clipped. The frames in grazed areas were placed at 15.2 m (50 feet) and 30.5 m (100 feet) along five transects with headings of 72 degrees apart radiating outward from the cage. All herbaceous species in each sample plot were collected. This avoided the uncertainty of collecting only certain forage species which may be difficult to identify when grazed and may not be representative of the community as a whole. Samples were air dried and weighed to the nearest 0.1 gram.

In riparian sites after the end of the grazing season, a 0.82 m² sample frame was used for plots inside the utilization cage and in two plots 15.2 m and 30.5 m upstream and downstream from the cage, for a total of four grazed plots at each location. Stubble heights (BLM 1996a) of Nebraska sedge were measured on a transect along the greenline, the first grouping of perennial vegetation along the water's edge (Winward 2000), in the vicinity of the riparian utilization cages.

Stubble heights were then correlated with paired plot utilization data.

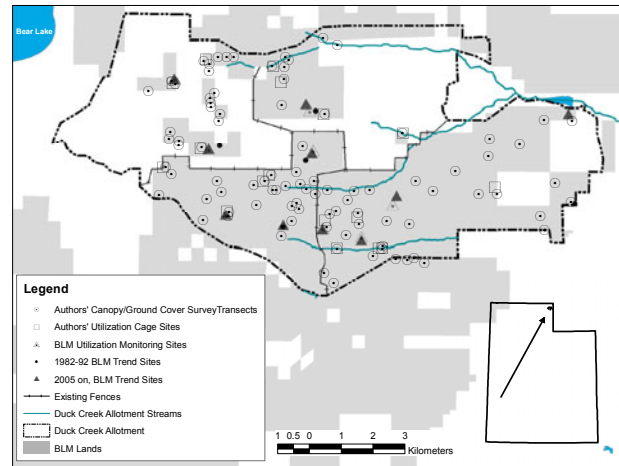


Figure 4. Location of authors' utilization and canopy cover survey sites.

Canopy and Ground Cover Surveys

In 2005, BLM conducted ecological site inventories (ESI) to describe the current status of the plant communities in terms of species, production and cover. The authors selected a number of sites that represented similar conditions found in representative BLM ESI locations where BLM also conducted rangeland health evaluations. The authors collected canopy and ground cover data (figure 4) for comparison to BLM data and to published canopy guidelines for sage grouse habitat (Connelly et al 2000). BLM data were collected in June and July, 2005. The authors' data were collected in May, June, July, September, and October 2008.

Ten sites were monitored from spring through fall in the south half of the allotment; an additional six sites in the north half were monitored during July. The quantitative line point transect intercept method (Herrick and others 2009) was used to collect canopy and ground cover. Radial transects (100' or 30.5 m) were placed in directions chosen from a random numbers table (Ott 1977). At each foot mark (0.3 m) on the tape, a metal pin was dropped through the vegetation layers and "hits" recorded for canopy of shrub, grass, forb and for grass >18cm and forb >18cm. Basal hits for bare ground, rock, crust, grass, forb, shrub and litter were also recorded. During the May and June samples, two transects at headings of 104° and 223° were surveyed for a total of 200 points at each location for each month. Two transects were added (at 241° and 289°), bringing the total points for each location to 400 for the July, September, and October surveys. This gave a total of 16,000 data points for these 10 transects (160 total transects) for

these five time periods. The July survey of the six additional locations in the north portion of the allotment recorded data from 24 transects and 2,400 data points.

Livestock Distribution and Census

The number of livestock that graze in an allotment, and the duration of grazing, are recorded by the grazing permit holder in "actual use reports." These can be validated but almost never are by field counts, including aerial surveys, of livestock (BLM 1984b). The authors counted the number of cattle grazing in the Duck Creek allotment during two aerial surveys conducted in 2006 and 2008. These used a fixed-wing aircraft traveling at approximately 150 km/h at an elevation of 250 m above the ground. A minimum of eight transects were flown. Where cattle were concentrated, quadrant surveys (circling of the aircraft) were conducted to note the location and number of cattle within each transect. The count at each location was checked a minimum of four times. Data were recorded on a field map and later entered in a GIS layer for display and tabulation.

Water Quality Monitoring

The authors sampled water quality in seven streams on BLM lands in Rich County during August, 2009. EPA-approved methods were used to monitor for key water quality parameters such as *E. coli*/fecal coliform, temperature, pH, dissolved oxygen, and turbidity. Streams monitored were Duck Creek, Six Mile Creek, and the North Fork of Sage Creek. A Hach HQ20 Portable LDO Dissolved Oxygen meter was used at each site to collect water temperature and dissolved oxygen data. As a quality control check, additional readings for temperature (water and air) were taken with an H-B Instrument Co. Enviro-Safe thermometer. A Hach 2100P turbidity meter was

used to measure sample turbidity for each site visit. A Hach SensION2 portable pH/ISE meter was used to measure pH. A Garmin eTrex GPS unit was used to collect location data in latitude and longitude at each site. The *E. coli*/Fecal coliform analyses were conducted using IDEXX Laboratories equipment to run Colilert® tests for each sample. The equipment set includes a Quality Lab Model WW-64835-00 Incubator, the IDEXX Quanti-Tray® Sealer Model 2X, sealing tray(s), Quanti-Tray® 2000 cards, ampoules of Colilert® reagent, a Spectroline EA-160 ultraviolet lamp for *E. coli* delineation, and 100ml Whirl-Pak® bags to collect samples. Samples were diluted 10:1 for streams with expected high coliform concentrations. Temperature, pH, dissolved oxygen, turbidity, and *E. coli*/fecal coliform were sampled 5 times within 30 days (separated by at least 3 and no more than 7 days between samples) to allow calculation of a monthly geometric mean for *E. coli* at each site.

BLM's Data Collection Methods

Utilization Data Collection

BLM conducted utilization monitoring from 2005 to 2008 using the key species method (BLM 1996a). This qualitative assessment uses an ocular estimate of the amount of forage removed by weight on an individual key species plant. Examiners walk along a transect and estimate the amount of utilization based on descriptions found in table 2. This method recommends that an ungrazed reference area be available for comparison. Training of observers involves comparison of estimated utilization with clipped and weighed sample plots. Utilization monitoring typically is a qualitative measure of the general appearance of a few key species.

Table 2. BLM qualitative key species method utilization classification system.

Utilization Class	Class Description
0-5% utilized	"the key species show no evidence of grazing use or negligible use"
6-20%	"the key species has the appearance of very light grazing. Plants may be topped or slightly used. Current seed stalks and young plants are little disturbed"
21-40%	"the key species may be topped, skimmed, or grazed in patches. Between 60 and 80 percent of current seed stalks remain intact. Most young plants are undamaged"
41-60%	"half of the available forage (by weight) on key species appears to have been utilized. 15-25 % of current seed stalks remain intact"
61-80%	"more than half of the available forage on key species appears to have been utilized. Less than 10% of the current seed stalks remain. Shoots of rhizomatous grasses are missing"
81-94%	"the key species appears to have been heavily utilized and there are indications of repeated use. There is no evidence of reproduction or current seed stalks"
95-100%	"the key species appears to have been completely utilized. The remaining stubble is utilized to the soil surface"

Source: BLM. 1999. Technical Reference 1734-3, Utilization studies & residual measurements, key species method, pages 81-85.

Ecological Site Inventories

Ecological site inventories collect data including plant species and productivity. When these data are compared with the plant community at its ecological potential, a similarity index can be determined (Habich 2001). The similarity index is calculated by comparing the occurrence of plant species for a sample site to reference areas or to the Ecological Site Type description (NRCS 2009).

In 2005, BLM conducted ecological site inventories in the Duck Creek Allotment to use in BLM's rangeland health assessments (BLM 2001a). See figure 5. BLM's purpose in using the ecological site inventory was to compare the composition and production of plant communities found today with the appropriate ecological site at its potential. This survey method, which involves estimating the amount of annual production (air dry weight) for each species observed along sample transects, is used to calculate a similarity index. The species production is used to calculate the similarity of the sample site with the plant community for this ecological site in climax condition. The annual production for the species identified is summed and compared with a similar sum for the climax community.

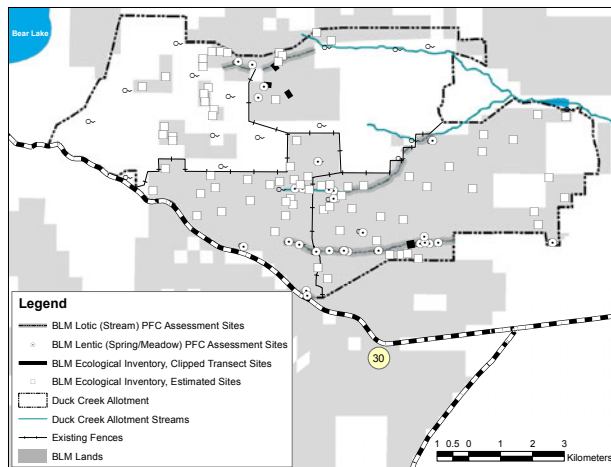


Figure 5. Location of BLM rangeland health and riparian properly functioning condition assessment sites.

BLM used double sampling (BLM 2001a) to collect data at four transects on the Duck Creek Allotment. Each transect had 20 plots where annual production by species was estimated. Two plots on each transect were clipped and weighed wet and then compared to an estimate for annual production that BLM made on the same transect for that plot. Comparison of clipped

and estimated values led to a correction factor, which was then applied to the 20 estimated plots on the transect. Assumed corrections were then applied to the field data to: 1) convert the weight of green clipped plants to air dried weight; 2) adjust for the amount of utilization that occurred prior to sampling; and 3) adjust for the percent growth when sampling early or midway through the growing season. The corrected data for all species BLM sampled were then totaled and that total compared against a total for a climax community. The resulting similarity index, expressed as a percent, was then ranked in one of four successional stages: 0-25 percent early; 25-50 percent mid; 51-76 percent late; and 77-100 percent potential natural (climax) community (BLM 2001a).

For the Duck Creek Allotment, BLM concludes that a similarity index of 50 percent or better is ranked as "functional" and meets rangeland health standards (BLM 2008b). BLM used the similarity index results as a key factor to assess whether rangeland health standards were met on the allotment.

Trend Data Collection

Collection of trend data as practiced by BLM (BLM 1996b) typically includes measuring the frequency of key plant species along a transect. Holechek and others (2004) recommend measuring trend at intervals of ≤ 5 years. In the case of Duck Creek BLM has measured trend at intervals between 2 and 12 years, using different locations; this makes analysis of trends at a site impossible. Trend data are considered inadequate to assess whether rangeland health standards are being met (Pellant and others 2000).

From 1962 to 2007, trend data were collected by the BLM at a number of sites using different methods (Figure 4). From 1962 to 1979, the photo plot method (BLM 1985) was used at two sites; from 1982 to 1992, the nested frequency sampling method (BLM 1985) was used at five sites; in 2004, an unknown method was used at a new site; and from 2005 to 2007, the line point intercept method (BLM 1985, Herrick and others 2009) was used at ten new sites.

Rangeland Health Assessments

BLM assessed rangeland health in 2005 at 34 sites. At each of these 34 sites, BLM scored 17 qualitative indicators of soil stability, hydrologic function, and the integrity of the biotic community at an ecological site level (Pellant et al 2000).

Riparian/Wetland Assessments

BLM's rangeland health assessments for riparian areas were based primarily on Properly Functioning Condition assessments for lotic and lentic areas (Prichard 2003a, Prichard 2003b). A properly functioning stream, or lotic area, has stabilized banks to dissipate high water flows in a manner that prevents unwanted erosion, traps sediment, and supports floodplains (BLM 1998). A properly functioning lentic area (springs, ponds, and meadows) has stability due to plants, which prevent excessive erosion, trap sediment, and support ground water recharge (Prichard 2003). The Duck Creek Allotment has more than 13 km of streams and 29 springs and wet meadows. Beginning in 2001, BLM assessed 29 lentic sites and 14 stream segments for properly functioning condition (figure 5).

RESULTS

Herbaceous Plant Annual Production and Utilization

From 2005 through 2009, each year the authors collected paired plot samples of herbaceous residual

vegetation in 670 sample plots for a total of over 1,300 samples for grasses and forbs. The residual vegetation found inside the utilization cages represents growing season production protected from grazing for both upland (table 3) and riparian areas (table 4). In 2005 seven upland sites were surveyed. From 2006 to 2009, twelve upland sites were surveyed. Table 5 compares measured upland grass production to the production predicted to occur on specific range sites, as described by the Rich County Soil Survey (SCS 1982; NRCS 2009). Values ranged from 25 to 76 percent of potential.

Upland grazing utilization measured by paired plots from 2005 to 2009 (based on grass and forb residual weights in grazed areas compared to ungrazed utilization cages) is described in table 6. Utilization ranged from 0 to 87 percent. In 2007, BLM personnel visited seven of the authors' upland sites where they measured utilization using the key species method. BLM's and the authors' results are compared in table 6. BLM's utilization results were consistently lower than the authors'.

Table 3. Duck Creek allotment herbaceous plant production in kg/ha in upland areas, based on the Authors' paired plot data.

Site	2005	2005	2006	2006	2007	2007	2008	2008	2009	2009	Averages	
	Grass	Forb	Grass	Forb	Grass	Forb	Grass	Forb	Grass	Forb	Grass	Forb
U1	48	275	99	173	304	38	272	201	175	114	180	160
U2	307	188	232	19	288	2	132	115	153	109	223	87
U3	112	229	135	37	226	0	86	82	87.1	104	129	91
U4	213	302	169	42	168	78	278	148	150	62	196	126
U6	304	417	350	145	190	62	208	242	186	238	248	221
U8	218	100	210	102	345	84	445	117	301	358	304	152
U9	207	130	191	6	135	1	323	25			215	41
U11			59	36	363	4	801	121	739	28	492	47
U12			183	169	353	146	411	285	350	205	325	202
U13			198	507	132	26	445	84	124	293	226	228
U14			67	165	174	6	87	177	108	134	109	121
U15			44	7	150	2	125	85	242	26	141	30
Average	202	234	161	117	236	38	302	140	238	152	232	126
SD	95	109	87	139	89	46	206	74	185	108		

Note: all data collected underneath grazing utilization cages thus protected from grazing.

Table 4. Duck Creek allotment herbaceous plant production in kg/ha for riparian areas, based on the authors' paired plot data.

Riparian Site	2005	2006	2007	2008	2009
U5	1,883	955	1,264	1,263	1,988
U7	1,013	419	900	1,667	482
U10	1,975	404 ^a	2,038	1,684	N/A ^b

^aUtilization cage U10 damaged, clipped small area remaining. ^bUtilization cage U10 damaged and no ungrazed residual vegetation to clip. Note: all data collected underneath grazing utilization cages protected from grazing.

Table 5. Grass annual production by range site based on authors' data for the Duck Creek Allotment.

range site ^a	ha	% of area	ha <50 % slope	authors' sites	grass production avg. of authors' sites kg/ha	grass production potential by range site, at normal precip. year ^b kg/ha	area accessible to livestock in the interspace between shrubs ha	total accessible grass production, authors' 2006-2009 data kg	total accessible grass production at potential for a normal year kg
Mountain Loam	14	(<1%)	12						
Semidesert Loam	2591	28%	2584	U2, U3	167	428	1,731	289,077	740,868
Semidesert Stony Loam	932	10%	929	U4	158	423	622	98,898	263,106
Upland Loam	2016	22%	1986	U1, U6 U13, U14	195	792	1,331	259,545	1,054,152
Upland Shallow Loam	2353	26%	2314	U8, U9 U11, U15	293	856	1,576	461,768	1,349,056
Upland Shallow Loam (Juniper)	132	1%	95			720			
Upland Stony Loam	1157	13%	1099	U12	324	428	736	238,464	354,752
Woodland	0	(<1%)	0						
Not Identified	4	(<1%)	0						
Totals	9199	100%	9018					1,347,752	3,761,934

^aBLM 2004 Duck Creek Project EA UT-020-2004-0030

^bSCS 1982 Rich County Soil Survey

We assessed grazing utilization in three riparian sites on the Duck Creek allotment using paired plots (table 7). At each site stubble height of Nebraska sedge was measured. Table 7 reflects the relationship between Nebraska sedge stubble height and grazing utilization at these sites. Stubble heights were measured at 7 additional sites to determine if the stubble height data at the three sites were comparable to other grazed riparian areas (table 8). The BLM requires that stubble heights be more than 12.7cm at the end of the grazing season. Data in table 8 reports that stubble heights of Nebraska sedge were less than 12.7 cm with utilization ranging from 85.7 to 97.4 percent. During August 2005, one month prior to the end of cattle grazing season, stubble height of grasses at two sites in meadows adjacent to Duck Creek and along the greenline were measured and compared. Meadow stubble heights at two different sites (RS1

and RS2) were 3.4 cm and 4 cm compared to greenline stubble heights of 8.1 cm and 6.1 cm respectively. Riparian utilization away from a stream was found to be higher than that measured along the greenline.

Canopy and Ground Cover

Table 9 provides a summary of the mean canopy cover for 10 sites located in BLM ecological sites used in rangeland health assessments in the south half of the allotment. Means were calculated across all sites within each month. Total shrub canopy (sagebrush, rabbitbrush, snowberry and others) averaged 33.3 percent. Shrub canopy other than sagebrush varied from 0.5 to 9.5 percent and averaged 5.1 percent. The overall sagebrush canopy averaged 28.2 percent. BLM estimated sagebrush canopy for the allotment as 38 percent.

Table 6. Upland percent grazing utilization, Duck Creek allotment, authors' and BLM data, 2005-2009.

BLM Site	Author Site	Separation Meters	BLM Sites Species Assessed	Author Sites	2007 BLM Assessment at Authors Sites, Species Assessed
DC 1	U8	133m	2005:STLE 11%* 2007:STLE 37%, POA 36% 2008:STLE 42%, POA 44%	2005:10%* 2006:71% 2007:61% 2008:81% 2009:87%	
DC 2	U6	256m	2005:STLE 16%, POA 12%* 2007:STLE 26%, POA 25% 2008:STLE 42%, POA 37%	2005:53%* 2006:65% 2007:67% 2008:71% 2009:81%	
DC 3	U9	487m	2005:STLE 12%* 2007:STLE 11%, PONE 8%, AGSP 12% 2008:STLE 29%, PONE 33%, AGSP 27%	2005:27%* 2006:20% 2007:0% 2008:49% 2009:na	STLE 21 % POFE 22%
DC 4		na	2005:STLE 13%, POA 17% 2007:STLE 34%, POA 32% 2008:STLE 28%, POA 20%		
DC 5	U1	134m	2005:AGSM 18%, AGSP 21% 2007:AGSM 19%, AGSP 21%, POA 17% 2008:AGSM 24%, AGSP 27%, POA 23%	2005:54% 2006:71% 2007:80% 2008:54% 2009:63%	STLE 20% POFE 30%
DC 7	U2	256m	2005:PSSP 17%, POA 23% 2007:POFE 32%, AGSM 30% 2008:POFE 9%, AGSM 8%, AGSP 10%	2005:75% 2006:73% 2007:84% 2008:0% 2009:56%	STLE 23% POFE 30%
DC 8		na	2005:POA 31%, PSSP 30% 2007:POA 29%, PSSP 25% 2008:POA 15%, PSSP 18%		
DC10		na	2005:STLE 30%, POA 40% 2007:STLE 27%, POA 30% 2008:STLE 6%, POA 6%		
	U3	na		2005:68% 2006:51% 2007:80% 2008:23% 2009:27%	STLE 26% AGSP 15%
	U4	na		2005:40% 2006:10% 2007:54% 2008:57% 2009:44%	STLE 18% AGSP 13%
	U11	na		2006:3% 2007:63% 2008:78% 2009:87%	POFE 40% AGSP 26%
	U12	na		2006:62% 2007:79% 2008:77% 2009:77%	STLE 37% PONE 32%
	U13	na		2006:76% 2007:0% 2008:71% 2009:75%	
	U14	an		2006:76% 2007:71% 2008:38% 2009:52%	
	U15	na		2006:10% 2007:78% 2008:46% 2009:80%	

***Bolded** text means that the site was rested from grazing during that year.

Table 7. Stubble height of Nebraska sedge compared to percent utilization in Duck Creek allotment riparian sites.

Location	2005	2006	2007	2008	2009
Duck Creek (U5)	8.1 cm (85.7%)	8.3 cm (79.1%)	9.0 cm (96.4%)	7.9 cm (94.8%)	6.7 cm (97.4%)
Six Mile Creek (U7) ^c		9.1 cm (87.2%)	7.6 cm (90.8%)	<10 cm (95.3%)	5.0 cm (96.9%)
S. Fork Six Mile Creek (U10) ^c		7.5 cm (93.7%)	8.0 cm (96.6%)	<10 cm ^a (97.3%)	5.4 cm ^b

^aAuthors' observations for stubble height. ^bThe two cages at site U10 in 2008 and 2009 were turned over and utilization could not be measured. ^cIn 2005 sites U7 and U10 were no grazed.

Table 8. Nebraska sedge stubble height (cm) measurements taken at authors' Duck Creek Allotment riparian monitoring sites (U5, U7, and U10) at the end of grazing season, along with seven other sites in watershed, 2005 – 2009.

Location	2005	2006	2007	2008	2009
Duck Creek (U5) (RS1)	8.1 (2.5) ^a	8.3 (3.0)	9.0 (3.6)	7.9 (2.4)	6.7 (3.3)
Duck Creek (RS2)	6.1 (1.6) ^a		5.0 (1.6)	5.5 (1.8)	4.1 (1.7)
Duck Creek Red Spring			7.6 (2.1)	6.7 (2.1)	4.0 (1.4)
Duck Creek Rich Spring				9.7 (0.7)	6.3 (2.5)
Six Mile Creek (U7)		9.1 (2.7)	7.6 (3.8)	<10 ^b	5.0 (1.5)
S. Fk Six Mile Creek (WP123)				7.8 (2.9)	6.1 (2.6)
S. Fk Six Mile Creek (WP124)				5.8 (1.7)	6.8 (2.4)
S. Fk Six Mile Creek (WP125)				5.8 (1.9)	4.8 (1.6)
S. Fk Six Mile Creek (WP126)				7.6 (2.3)	6.6 (1.9)
S. Fk Six Mile Creek (U10)		7.5 (2.7)	8.0 (2.3)	<10 ^b	5.4 (2.2)

^aMeasured one month prior to the end of the grazing season. ^bAuthors' observation. Parenthesis denote standard deviations.

Table 9. Average canopy cover percent measured by authors at BLM Ecological Sites in Duck Creek Allotment.

Month	Total Shrub	Total Grass	Total Forb	Grass >18cm high	Forb >18cm high
May	31.7 (4.1) ^a	7.2 (4.1)	3.1 (1.6)	0.0 (0)	0.0 (0)
June	34.8 (6.4)	17.6 (3.7)	15.0 (4.1)	4.5 (2.5)	1.2 (2.1)
July	33.6 (5.3)	18.7 (3.2)	12.4 (4.9)	5.7 (2.7)	2.2 (1.6)
September	33.4 (5.1)	17.4 (3.0)	9.1 (4.7)	2.9 (1.6)	1.2 (1.0)
October	33.1 (6.0)	19.0 (3.4)	9.2 (4.7)	2.1 (1.2)	0.6 (0.5)
Overall	33.3 (5.3)	16.0 (5.6)	9.7 (5.7)	3.0 (2.7)	1.0 (1.4)

^aNumbers in parenthesis are the standard deviation.

Table 10. Average ground cover percent for ten BLM Ecological Sites on Duck Creek allotment.

Month	Bare Ground	Rock	Crust	Grass ^a	Forb ^a	Shrub ^a	Litter
May	25.6 (6.6) ^b	4.1 (2.9)	2.8 (1.5)	4.5 (1.6)	9.0 (3.9)	0.8 (0.9)	53.3 (7.7)
June	23.2 (5.8)	3.8 (3.3)	1.8 (1.2)	3.6 (1.3)	6.1 (4.9)	1.3 (0.9)	60.4 (6.8)
July	23.9 (5.5)	4.2 (3.8)	0.9 (0.9)	3.3 (1.0)	4.8 (2.9)	2.5 (1.0)	60.4 (6.2)
Sept	22.8 (7.0)	4.3 (3.0)	0.9 (0.9)	3.4 (0.9)	2.4 (1.6)	3.5 (1.0)	62.8 (7.9)
October	23.6 (9.3)	2.4 (2.4)	1.1 (0.7)	3.1 (0.5)	1.1 (0.7)	3.4 (0.6)	65.4 (10.4)
Overall	23.8 (6.7)	3.7 (3.1)	1.5 (1.3)	3.6 (1.2)	4.7 (4.1)	2.3 (1.4)	60.5 (8.6)

^aIncludes basal hits on shrubs at ground level. ^bNumber in parenthesis are the standard deviation.

The authors found that shrub canopy remained consistent through the seasons. Total canopy of grasses and forbs increased from spring into summer to a maximum of 19 and 15 percent, respectively. Grasses and forbs >18 cm in height increased from spring to summer and then decreased into fall with

maximum grass canopy of 5.7 and forb of 2.2 percent. The authors' ground cover measurements at different times of the year are summarized across these ten ecological site locations in table 10. Average ground cover values for the five sample periods were: bare ground (23.8 percent); rock (3.7

percent); crust (1.5 percent); grass (3.6 percent); forbs (4.7 percent); shrubs (2.3 percent); and litter (60.5 percent). These averages remained consistent over the months with only the forbs showing a gradual decline from the spring through the fall.

Comparisons of BLM canopy and ground cover estimates (BLM 2008b) with the authors' 2008 data are shown in tables 11, 12 and 16. Authors' measurements of canopy cover (table 11) showed variation within sites for shrubs and forbs, with BLM reporting higher canopy cover of shrubs by 3 percent, grasses by up to 9 percent more, and forbs less by 1 percent. Table 12 shows BLM survey estimates for

litter, bare soil, and rock which BLM combined together. If the authors' bare ground, rock and litter data are combined, on average the authors found this total to be three times more than BLM reported. The differences in methods (BLM's subjective estimate versus the authors' line point intercept data) may explain why more bare ground and litter amounts were measured by the authors. Table 16 presents the authors' ground cover data in two categories, under shrubs and between shrubs where we summarize the fraction of ground cover for bare ground, rock, biotic crust and plants that were under shrubs or in the inner space between shrubs.

Table 11. Comparison of BLM canopy estimate and authors' data at 10 BLM ecological inventory sites in the Duck Creek Allotment.

BLM Site	Shrub %		Forb %		Grass %	
	BLM	Authors	BLM	Authors	BLM	Authors
DC7	30	38.4	10	9.6	30	9.6
DC8	30	37.7	10	12.6	30	18.3
DC9	30	37	10	11.7	35	13.9
DC10	45	27.1	10	11.2	20	15.8
DC11	45	30.2	5	7.0	20	14.9
DC11(a)	45	35	5	4.1	20	17.2
DC17	40	31.1	5	9.2	25	12.9
DC19	15	24.8	10	5.8	35	18.9
DC25	40	36.5	6	12.4	18	18.9
DC26	45	35.5	10	13.9	20	19.6
Average	36.5	33.3	8.1	9.75	25.3	16.0

Table 12. Comparison of BLM ground cover percent estimates and authors' ground cover data at ten BLM Ecological Sites in Duck Creek allotment.

BLM Site	BLM Data	Authors Data						
	Litter, Bare, Rock %	Bare	Rock	Crust	Grass	Forb	Shrub	Litter
DC7	L+B+R = 30	27.1	2.6	1.6	2.7	3.4	2.1	60.5
DC8	L+B+R = 30	20.6	2.4	5.5	3.7	7.3	2.6	63.0
DC9	L+B+R = 25	17.8	0.2	0.8	4.0	5.4	2.8	69.0
DC10	L+B+R = 25	28.1	2.0	3.2	4.2	3.4	2.3	57.1
DC11	L+B+R = 30	20.5	4.4	2.1	4.7	3.5	2.5	62.4
DC11(a)	L+B+G = 30	35.9	9.0	1.9	3.6	1.4	2.3	46.0
DC17	L+B+R = 30	22.2	1.0	1.8	3.7	3.3	2.1	66.0
DC19	L+B=40	29.2	8.5	1.5	3.1	3.2	2.0	52.7
DC25	L+14B=36	17.8	2.4	0.5	3.6	7.0	2.1	66.8
DC26	B+R=25	18.7	5.2	1.0	2.5	9.2	2.4	61.3

L litter, B bare ground, R rock

Table 16. Comparison of ground cover percent total, beneath under shrubs and inter space between shrubs.

	Bare Ground	Rock	Crust	Grass, basal	Forb, basal	Shrub	Litter
Total	23.8 (6.7) ^a	3.7 (3.1)	1.5 (1.3)	3.6 (1.2)	4.7 (4.1)	2.3 (1.4)	60.5 (8.6)
Beneath shrub	3.1 (2.0)	0.3 (0.4)	0.7 (0.9)	1.1 (0.7)	1.7 (1.8)	0.0 (0.0)	26.3 (5.1)
Inner space	20.7 (5.9)	3.4 (2.8)	0.8 (0.9)	2.4 (0.8)	3.0 (2.6)	2.3 (1.4)	34.1 (6.5)
Shrub/Total %	13.0	9.0	45.8	31.6	36.9	0.0	43.6

^aStand deviation is shown in parenthesis.

BLM Ecological Site Inventory and Rangeland Health Assessments

In 2005, BLM collected field data using the ecological site inventory (ESI) method for use in determining whether rangeland health standards are being met on the Duck Creek allotment (figure 5). The ratings on the 28 sites in The Duck Creek Allotment for ESI indicators are displayed in Appendix A along with the ESI Similarity Index for that site for average and wet precipitation years. BLM also assessed the condition of seven stream segments and 28 springs and meadows in Duck Creek. The results of these Properly Functioning Condition (PFC) assessments are included in Appendix B (streams) and Appendix C (springs).

Water Quality Data

Rangeland health standards require that a stream meet state water quality standards (BLM 1997). The results of data collection by the authors in 2009 for six criteria for Utah water quality standards are described in table 13. Water temperature exceeded state criteria in Duck, North Fork Sage, Sage, and South Fork Six Mile Creeks, while it remained below criteria in Big,

Otter, and Randolph Creeks. Measured pH at each sampled stream was generally within the criteria range, although small exceedances were found in North Fork Sage Creek and Sage Creek. Dissolved oxygen in all streams met criteria. While industrial emissions need to meet turbidity requirements, nonpoint sources which cover agricultural practices such as domestic livestock grazing do not have a turbidity standard. However, the authors did measure turbidity in the field. Turbidity values in all streams experienced highs that were several times higher than their lows, or background levels, during the five sampling episodes. Observations during sampling showed that instream disturbance and bank trampling of eroding stream banks by cattle lead to increases in sediment and turbidity. The *E. coli* geometric mean concentrations at the sampled sites exceeded the Utah water quality standard in Big, Duck, North Fork Sage, Randolph, Sage, and South Fork Six Mile Creeks. The Otter Creek geometric mean (195 MPN/100 ml) was near the state criterion of 206 MPN/100 ml. Maximum *E. coli* levels found in all streams exceeded the Utah maximum criterion of 668 MNP/100 ml for single readings.

Table 13. Water quality data in Duck Creek Allotment streams and other nearby streams.

Location	Mean Water Temp °C ¹	Mean pH Units ²	Mean Dissolved Oxygen ³ mg/l	Nonpoint Source Mean Turbidity ⁴ NTU	E.coli Range ⁵ MPN	Geometric Mean E.coli ⁶ MPN/100 ml
Big Creek	11.5	8.4	10.4	4.9	119-1,203*	360*
Duck Creek	22.3*	8.0	7.0	49.3	2,481-12,997*	2,719*
N. Fork Sage Creek	20.0	8.3	7.1	588.4	14,136->24,196*	5,103*
Otter Creek	15.2	8.4	8.0	2.7	81.6-727*	195
Randolf Creek	13.8	8.4	8.8	5.4	1,046-2,420*	1,600*
Sage Creek	21.0*	8.5	7.4	317.6	3,654-19,863*	2,974*
S. Fork 6 Mile Creek	23.0*	8.1	7.1	69.5*	998-3,076*	239*

Utah water quality standards: ¹Temperature C maximum 20, ²pH range units 6.5-9.0, ³Dissolved oxygen minimum 30 day average mg/l <6.5, ⁴Turbidity increase NTU for point sources [10] - (there is no nonpoint source standard for turbidity), ⁵E. coli maximum number / 100 ml <668, ⁶E. coli geometric mean, number /100 ml <206. * Values where Utah water quality standards were not met.

Table 14. Number of cattle surveyed on allotment compared to the number reported by rancher and number permitted on Duck Creek and other BLM allotments.

BLM allotment	Year Surveyed	Field survey (# cattle)	Reported use (# cattle)	Permitted (# cattle)	% of reported	% of permitted
Upper Cattle, GSENM ^f , UT ^a	2007	222	774 ^c	1093	29%	20%
Alvey Wash, GSENM ^f , UT ^a	2009	65	295 ^e	252	22%	26%
Lower Cattle, GSENM ^f , UT ^a	2009	364	614 ^d	1284	59%	28%
Vermillion, GSENM ^f , UT ^b	2007	33	140 ^c	281	24%	12%
40 Mile Ridge ^b , GSENM ^f	2008	183	480 ^e	570	38%	32%
Smiths Fork, WYO ^a	2008	439	1449 ^d	2146	30%	20%
Duck Creek, UT ^a	2006	450	641 ^{c, d}	641	70%	70%
Duck Creek, UT ^a	2008	304	641 ^{c, d}	641	47%	47%
Duck Creek, UT June 25 ^a	2010	570	641 ^d	641	89%	89%
Duck Creek, UT Sept 4 ^a	2010	148	641 ^d	641	23%	23%

^aAerial survey, ^bGround survey, ^cFrom permittee supplied "Actual Use Reports", ^dFrom billing statements, ^eBLM estimated average over 10 years, ^fGSENM—Grand Staircase-Escalante National Monument.

Livestock Census and Distribution

Aerial surveys were conducted in 2006 and 2008 to determine the distribution and number of cattle within the Duck Creek Allotment (table 14). In 2006, 450 mature cattle were counted, 85 percent of which were in the northwest pasture on June 26. In 2008, 304 were counted, 95 percent of which were located in the northeast pasture on June 24. In 2006, 2008, and 2010 BLM reported grazing billing for 641 cow-calf pairs to graze in the Duck Creek Allotment. The permit holder's actual use reports for 2006 and 2008 reported the same numbers.

DISCUSSION

Ecological Indicators, Policy Assessment, and Determination of Whether Standards Met

This discussion reviews the relationships among ecological condition indicators, ecological goals, standards, and assessment methods in the context of data collected for the Duck Creek Allotment. Our independent assessment of the ecological conditions on the Allotment is discussed in terms of causal factors of specific habitat conditions and potential management changes to reduce undesirable stressors.

Ecological condition indicators include species composition and diversity, biomass (or net primary production), nutrient stock, and ecosystem structure and processes (Westman 1978). The number of trophic levels and whether species are genetically linked through habitat connectivity are also included as indicators of ecosystem conditions (Montoya and others 2006). Conditions measured by each of these indicators are important over time (Soulé 1985) and at

different geographic and spatial scales (Scott and others 1999). When habitat resilience is diminished, disturbance can cause the system to cross a threshold to a new ecological state from which recovery is sometimes not possible (Groffman and others 2006). To prevent a transition to an undesired state, land managers must know where state change threshold occurs, what stressors will cause the system to cross the threshold, and the kind of control of stressors needed to prevent crossing the threshold (Thrush and others 2009, Miller 2005). The concept of states and thresholds is largely conceptual and has yet to be defined empirically, and so is difficult to integrate into land management. In the meantime, management that insures resiliency and ecological capacity (e.g., managing for protected core areas, landscape connectivity, key species viability, and biodiversity) is recommended (Cumming and others 2005). Inherent to this process is restoring and sustaining the productivity of native ecosystems.

BLM's range management program makes ecological assessments to determine whether standards for habitat are met. A number of field assessment methods have been developed by BLM. Do these assessment methods provide the kind and quality of information needed to assess ecological indicators? Table 15 compares this simplified set of ecological indicators to the methods used by BLM: trend, utilization, Ecological Site Inventory, upland rangeland health assessments, and riparian ecological health assessments. We reviewed each of these methods, their application, and their utility in assessing resilient habitat. Using the criteria described in BLM's Handbook 4180, we reviewed the primary

assessment methods BLM uses to determine whether they: 1) are relevant to the specific standard(s); 2) manage for responses that are detectable; 3) describe the minimum suite of indicators needed; 4) provide results that are credible among a diverse audience; 5) use methods that are standardized and accepted; and 6) can distinguish between whether an indicator does or does not meet standards (BLM 2001b). The ecological indicators (rows in table 15) reflect vital signs of ecosystems that are practical to measure (Kurtz and others 2001). These vital signs are chosen to reflect the key natural elements and processes (primary production, trophic transfer, nutrient cycling, water dynamics, and energy transfer) in ecosystems (Miller 2005). Table 15's ecological indicators for biological processes emphasize measures for biodiversity such as species richness, evenness, disparity, rarity, and genetic variability. This indicator is further broken down into additional important biological processes. Each assessment method in table 15 was evaluated on how completely its use would assess the ecological indicators. The results (yes, limited, no) indicate how comprehensive the assessment method is to evaluating ecological health. A majority of the ecological indicators in table 15 are not assessed by the current assessment methods assigned by BLM for this task. Many of the assessment methods offer limited ability to measure the ecological indicator. Only two of the assessment methods seem adequate for two ecological indicators.

Rangeland Evaluations

Trend

Trend and similarity index data were used by BLM to assess whether rangeland health standards are being met in the Duck Creek Allotment (BLM 2008b). Trend data from the earliest monitoring (1969 to 1979) in the Allotment has been lost. Based on data collected at five sites in the Allotment from 1982 to 1992, BLM concluded that the trend was up at four sites and static-to-down at the fifth site (BLM 2008b). The data from this period (1982-92) show significant increases in western yarrow, rabbitbrush, sagebrush, and spiny phlox, all of which are grazing tolerant species that increase with livestock grazing. During this same period, declines were seen in western wheatgrass and clover. A number of grass species persisted in trace amounts, including bluebunch wheatgrass and Sandberg's bluegrass. BLM data show that the trend is down for species livestock prefer and up for species livestock do not prefer. For instance, these data indicate low amounts of bunchgrass species such as bluebunch wheatgrass and Indian ricegrass, which should dominate these range sites but which are favored by livestock. Ecological condition assessments indicate that the trend is moving further away from potential native climax communities.

Because BLM has lost knowledge for the locations of these earlier trend sites (BLM 2008b), BLM established 10 trend sites at new locations in 2004.

Table 15. Evaluation of BLM range management policies and ecological assessment methods that represent ecological indicators.

		Assessment methods				
Ecological indicators	BLM rangeland health standards	Trend	Utilization	Ecological site inventory	Interpreting indicators for rangeland health	Lotic / lentic PFC
Soil nutrient processes	Std 1	No	No	No	Yes	No
Hydrological processes	Std 2,4	No	No	No	Limited ^c	Yes
Biological processes	Std 2,3	Limited ^a	No	Limited ^e	Limited ^c	No ^d
Plant community composition	Std 3	No	No	No	No	No ^d
Habitat structure	Std 2,3	No	No	Limited ^b	Limited ^c	Limited ^d
Habitat connectivity	Std 3	No	No	No	No	No ^d
Wildlife populations	Std 3	No	No	No	No	No
Are the above indicators considered in appropriate spatial scale?	n/a	No	No	No	Limited ^c	Limited ^d
Are the above indicators considered in appropriate temporal scale?	n/a	Yes	No	No	No	No

^aTrend data collection, as normally practiced by BLM, is limited to the frequency of a few key plant species at sample intervals sometimes a decade long. ^bEcological Site Inventories focus on generating a similarity index which is outside common ecosystem metrics. ^cInterpreting Indicators for Rangeland Health uses measures of ecologically concepts that have not been independently validated. In practice, only the survey-site scale and not watershed or regional scales are normally considered. Other key factors, such habitat needs for avian and terrestrial wildlife are not adequately assessed. ^dLotic and Lentic PFC assessments focus on site stability and erosion. Similarly, other key factors such habitat needs for aquatic, avian, and terrestrial wildlife are not adequately assessed.

Using line point intersect transects, BLM collected canopy, ground cover, and species frequency data in 2004 and 2005 at these new sites. From these two years of data, BLM concluded that the trend was upward (improving) based on "canopy cover and species richness." Yet, the line point intersect data did not show significant changes in this one-year period for canopy cover or the number of species. BLM did not analyze the effect on these attributes of higher precipitation in 2005 compared to 2004. BLM's trend data fail to support the conclusions BLM made that the trend is static or upward on most monitoring sites.

Ecological Site Inventory

BLM calculated the ESI similarity index for 34 sites in the Duck Creek Allotment (Appendix A). Of 28 sites assessed, BLM found that 23 sites had a similarity index of 50 percent or more, reflecting what BLM describes as a good, or late seral, ecological condition; nine were classified as mid-seral, one as climax, and one was not determined (BLM 2008b). These results were based on data collected in 2005, which was an above average precipitation year and consequently an above average production year.

Conversion of field production data on species involves applying a number of correction factors to convert collected samples into adjusted production for an average year. There is a clear indication that validation in the field is needed. Calculations based on these combined correction factors lead to a total production for sites in the Duck Creek Allotment that is two times higher than predicted by the ecological site descriptions.

Additional problems exist with BLM's similarity calculations. Using BLM's data, similarity of grasses to the potential natural community was 39 percent, with many sites below 25 percent or in poor condition, while forb similarity was 37 percent, and shrubs were 80 percent of the production of expected native species. By design, the way the BLM calculates the similarity index masks the fact that herbaceous species are often depleted. In shrub dominated communities, the high annual production of shrubs is averaged with those for the grasses and forbs in calculating the similarity index. As a result, the depletion of the native herbaceous community is masked by averaging its production with woody plant production.

Further analysis of BLM's ESI data reveals problems with native bunchgrasses such as bluebunch wheatgrass, which is a preferred livestock forage and the key species for the allotment. Bluebunch wheatgrass was found mostly in trace amounts at 13 of 28 BLM ESI sites. The Rich County Soil Survey (SCS 1982) indicates that this grass species should be dominant on the allotment. BLM data show that bluebunch wheatgrass annual plant production is present at 28 percent of the potential amount described in BLM's revised ecological site descriptions (NRCS 2005a, 2005b) or 12 percent of potential predicted in relevant soil-survey rangeland characteristics (SCS 1982). Indian ricegrass in 2005 was found at 10 of 28 sites and was present at 22 percent of potential described in the relevant ecological site description or 12 percent of potential described in the Soil Survey (SCS 1982). Because BLM's ESI data were collected in a wet year (2005), if adjusted for precipitation, the resulting percent of these species relative to their potential would be even lower. By any measure, because these dominant native bunchgrasses exist today at a fraction of their potential, this represents significant ecological deterioration.

The rhizomatous western wheatgrass, a grazing tolerant species, was present at 24 of 28 sites; the Soil Survey does not include it as an expected species present on this allotment for habitat conditions at ecological potential. Sandberg's bluegrass was present at 23 of 24 ESI sites and had the highest biomass of any grass on the allotment. Sandberg's bluegrass is grazing tolerant due to early maturation and short growth form. According to the Soil Survey, it should be present at only 11 of 28 ESI sites. It was present at 219 percent of potential. The plant community composition for the Duck Creek Allotment has shifted away from the potential plant community towards a community dominated by grazing tolerant species.

BLM has moved away from using the similarity index in assessing whether rangeland health standards are met. Interpreting Indicators of Rangeland Health, Technical Reference 1734-06 (Pellant and others 2000, Pierson and others 2002), is the primary method that BLM uses for rangeland health assessments in upland areas. The reference describes the problem with the similarity index and recommends not using it in determining if rangeland health standards are met.

The ESI procedure collects data on plant species and these species' estimated annual production at a site. While this is helpful, because it considers only plant taxa, it offers limited information on the wider array of animal and soil biota and we opine that it is not an appropriate method to use in order to assess ecological conditions and whether rangeland health standards are met. However, in the grazing renewal decision for the Duck Creek Allotment, BLM uses trend and similarity indices in making rangeland health determinations. As a result, those Duck Creek Allotment habitat areas with ecological problems were under reported by the BLM.

Upland Rangeland Health Assessments

The primary assessment method used by BLM to assess whether rangeland health standards are met is "Interpreting Indicators for Rangeland Health" (Pellant and others 2005, Pyke and others 2002). It's technical reference (TR1734-6) uses qualitative rankings of 17 indicators, which compare the survey site against a reference site that resembles the historic climax plant community for that ecological site type. The observer assigns one of five ratings to describe the deviation of the survey site from reference conditions. These rankings have limited relevance to ecological theory and, because they are subjective, are problematic to apply in the field.

Qualitative terms are linked to ecological condition in a way that makes it difficult to assess whether standards are met. The resulting determination of whether standards are met depends on a preponderance-of-evidence. In Utah, scores that are moderate in departure, slight to moderate, or slight to none are assumed to meet rangeland health standards (BLM 2008c). Only in cases where most of the indicators indicate extreme departure will the site be evaluated as not meeting rangeland health standards.

The results of the rangeland health assessments conducted by BLM found that 25 of the 28 upland sites evaluated in Duck Creek were "functioning" and therefore met standards, while 3 sites were functioning at risk.

One example of these indicators, that for bare ground, demonstrates the nature and limitations found with the other 17 indicators. The evaluation matrix for the bare ground indicator describes the departures from reference conditions for five rankings or scores: 1)

Extreme to total – "much higher than expected for site. Bare areas are large and generally connected." 2) Moderate to extreme – "moderate to much higher than expected for the site. Bare areas are large and occasionally connected." 3) Moderate – "moderately higher than expected for the site. Bare areas are of moderate size and sporadically connected." 4) Slight to moderate – "slightly to moderately higher than expected for the site. Bare areas are small and rarely connected." and 5) None to slight – "Amount and size of bare areas match that expected for the site."

Comparison of the survey site with a reference area is necessary to infer what is "expected for the site." Representative ecological sites that reflect ecological conditions at their potential are exceedingly rare on BLM lands. Without a representative reference area, there is a strong tendency to accept observed conditions as normal, therefore scoring them higher than they might deserve. For the surveys and assessments that BLM conducted in the Duck Creek Allotment in 2005 (sites 6, 7, and 8) no reference areas were used.

Indicators should predict biological community state transitions, particularly transition to a degraded state. Likewise, to document recovery, indicators should identify conditions that signal a positive change in state. TR 1734-6 cites numerous studies (Anderson 1974, Benkobi and others 1993, Cerda 1999, Gould 1982, Gutierrez and Hernandez 1996, Morgan 1986, Weltz and others 1998) which, while adequately describing ecological principles relating to bare ground, do not support the specific rankings used in TR 1734-6. Erosion that exceeds rates of tolerable soil loss over time will lead to state changes (NRCS 2010). The rangeland health standards call for soil stability that maintains soils at their ecological potential (BLM 1997). For the bare ground indicator, TR 1734-6 does not link the amount of bare ground for a survey site to the specific standard required for making an assessment. The assessment method fails to clearly link the relevant rangeland health standard to the assessment ranking and then support this with scientific studies.

As applied in the field, the amount of bare ground for the Duck Creek Allotment was not ranked as an ecological problem by BLM even though the authors' data showed otherwise. As described below, the authors' measurements of bare ground in the Duck Creek Allotment compared to reference areas show

significant departures from potential. Bare ground that the authors' measured in ungrazed reference habitat was extremely low. This suggests that the ranking for bare ground at most Duck Creek Allotment sites should have been "extreme to moderate" rather than "slight to none" departure from reference conditions.

The range site descriptions for the dominant soil types in the Duck Creek allotment identify cool season bunchgrasses as the dominant plant group for the allotment. Bluebunch wheatgrass, Nevada blue grass, needle and thread grass, and Indian rice grass should comprise about half of the annual plant production in these range sites. As described above, these cool season grasses are either absent or found in trace amounts in most range sites in the allotment today. Similarly, cryptobiotic crusts should be prevalent, particularly in the shrub interspace areas, but are rarely found in the line point transect data. The loss of this ecosystem component has far reaching ecological consequences in terms of wildlife support, nutrient flow, soil stability, and biodiversity. TR 1734-6 indicator 12 for functional and structural groups was rated "slight to none" or "slight to moderate" departure from reference conditions. The authors argue that the loss of key groups like cryptobiotic soils may justify a score of "moderate to extreme" departure. Similar arguments can be made for many other indicator ratings.

Spring and Riparian PFC Assessments

BLM relied primarily on lentic and lotic properly functioning condition assessments for evaluating health of riparian areas on the Duck Creek allotment. Of the 6 lotic and 29 lentic assessments, BLM found that 4 stream segments and 6 lentic sites are functioning at risk and thus not meeting rangeland health standards. The stream segments assessed in the Duck Creek Allotment are contained in narrow channels which have become incised or down cut by several feet and now are disconnected from their original, wide floodplains and riparian meadows. BLM's (1993) TR 1737-9 states that, "The absence of certain physical attributes such as a floodplain where one should be are indicators of nonfunctioning condition." This criterion does not appear in the later technical references used by BLM today (Prichard 2003b), and is no longer required in determining whether the streams are properly functioning.

Not all of the rangeland health standards are covered by the proper functioning condition assessments. For

example, Standard 2 requires that riparian areas have vegetation that provides "food, cover and other habitat needs of dependent animal species" such as fish. TR 1734-15 and TR 1734-16, which assess properly functioning condition of streams and springs, do not account for these requirements. Stevens and others (2002) describe some of the ecological shortcomings of TR 1734-15 and 1734-16.

For the Duck Creek Allotment, BLM determined that one of the six streams doesn't meet rangeland health standards and that livestock grazing is a factor (BLM 2008b). Additionally BLM reported that six of the 29 lentic locations surveys were functioning at risk and not meeting BLM's rangeland health standards. Based on a single assessment, BLM further noted that the trend for the riparian areas was "static or no apparent trend" toward potential.

Water Quality Assessments

BLM relied on Utah's 303d list of impaired waters to assert that water quality standards were met on the Duck Creek Allotment (BLM 2008b). However, these streams are not monitored by the State, and BLM did not conduct or have others conduct water quality surveys for the Allotment (BLM 2008b). Water quality data collected by the authors show that the sites sampled in Duck Creek fail to meet state temperature and *E. coli* standards (table 13). The elevated levels of water temperature, turbidity (sediment) and *E. coli* found in these streams are influenced by the presence of cattle in the streams and watershed. Activities affecting watersheds or riparian zones also affect stream ecosystems directly, indirectly, and cumulatively. Several reviews of livestock impacts on stream and riparian ecosystems have covered this topic in detail, using hundreds of government documents and peer-reviewed scientific articles. These include Kauffmann and Kreuger (1984), Armour and others (1991), Gregory and others (1991), Platts (1991), Fleischner (1994), and Belsky and others (1999). Livestock in the Duck Creek Allotment regularly trample, wade, defecate, and urinate directly in these streams causing fecal pollution, increased nutrient levels, algae blooms, increased sedimentation, and reduced dissolved oxygen, which impair habitat for native cutthroat trout and other native aquatic organisms. These conditions violate Utah's standards for water quality (Utah Administrative Code R317-2-7.2). These violations of Utah's water quality regulations would cause the streams on the Duck Creek Allotment to fail Standard

4 of the Utah Standards and Guidelines (BLM 1997) and, therefore, the fundamentals of rangeland health. BLM assumed that waters in the Duck Creek Allotment met rangeland health standards for water quality in the absence of water quality monitoring data.

Canopy Cover, Ground Cover and Sage Grouse Guidelines

While the standards and guidelines require vegetation necessary to ensure that native wildlife species populations are at their potential, the methods BLM uses for ecological assessments lack indicators for wildlife. Sage grouse is one of many "special status" species found in the Duck Creek Allotment, which BLM is obligated to consider in management decisions (BLM, 2008b). BLM (2008b) compared its estimates of cover by sagebrush, grasses, and forbs to the Connelly et al. (2000) guidelines for sage grouse habitat. The guidelines for spring nesting and early brood-rearing habitats are: sagebrush canopy of 15 – 25 percent; perennial grass canopy >15 percent for grasses >18 cm height; and forb canopy >10 percent for forbs >18 cm height. For summer brood rearing habitat, sagebrush canopy should be 10 – 25 percent with grasses and forbs >18cm height having a total canopy of >15 percent. Canopy of sagebrush in winter should range from 10 – 30 percent. Authors data (table 11) show that Connelly and others's criteria for grass canopy cover are met. However, the canopy for forbs, and the height required for grass and forbs was not met (table 7 and 9).

As reported above, the authors surveyed 10 of BLM's ESI sites during the spring nesting and early brood rearing period (May and June). None met the minimum sage grouse criteria for grasses and forbs >18cm in height. Of the 160 transects measured by the authors during the summer (July) and fall (September, October), 13 (8 percent) met the 15 percent total forb and grass cover with >18cm height. Eleven of these 13 transects were on steep slopes seldom grazed by cattle. The maximum canopy cover of grasses on these steep sites was 48 percent. 40 percent of sample points had grass over 18 cm in height. This high grass canopy on lightly grazed sites suggests potential for much higher canopy than that measured in most grazed sites and compares favorably with data from ungrazed kipukas in Wyoming big sagebrush communities in southern Idaho. In these kipukas, grass canopy ranged from 29 – 58 percent with an average canopy of 43.5 percent (Welch and Criddle 2003).

For most sites in the Duck Creek Allotment, BLM estimated the combined ground cover for bare ground, rock and litter at these sites, while not considering ground cover beneath shrub, forb and grass canopies (table 12). As a result BLM's data could not provide information which is important for erosion assessments and comparison to potential. Precipitation on the Duck Creek Allotment occurs mostly during the October – March period as snowfall. Summer rains are a small contribution to the total. Erosion is, therefore, mostly driven by overland flow from snowmelt, which is affected by overall ground cover rather than raindrop impact which is influenced by canopy cover. Bare ground under a shrub may be prone to water erosion while classified as covered by canopy cover. BLM did not measure ground cover beneath grass, forb, and shrub canopy, based on the assumption that canopy cover-intercepted rainfall is the most significant factor protecting the soil from erosion. West and Gifford (1976) found that shrub canopy cover intercepted about 1 percent of precipitation, refuting that canopy cover acts to protect ground cover from erosion. The authors argue that ground cover should be measured independently of canopy cover. When combined, bare ground under shrubs may be missed. For this reason, BLM's ground cover surveys are likely to under report the amount of bare ground.

By assessing what contacts the ground and not counting foliar or canopy cover as ground cover, the authors found that the average bare ground at surveyed locations was 25.3 percent (table 12), with most bare ground occurring in shrub interspaces where livestock access is not restricted (table 16). The authors surveyed an ungrazed highway right of way on the south side of the Duck Creek Allotment that had not been grazed by livestock for 30 years (UDOT 2009) and found that bare ground was 1 percent for this upland loam range site type, which is a dominant range site on the allotment. A study in the nearby Uinta-Wasatch-Cache National Forest in big sagebrush habitats where livestock had been excluded for decades measured 5.6 percent bare ground and 38.8 percent basal cover of grasses (Carter 2003). Thus these sites serve as reference areas. The Uinta-Wasatch-Cache National Forest provides ground cover values for various habitat types. In big sagebrush communities, the potential ground cover is 89 – 93 percent with a maximum of 96 percent (USDA Forest Service 2005).

The canopy and ground cover data just summarized provide yet another check in the overall ecological evaluation process. BLM did not consistently assess bare ground, which our data show is far from potential. The result is excessive erosion and the related rangeland health standard not being met. Sage grouse habitat needs are not built into the standard agency assessment process when determining whether rangeland health standards are met. Herbaceous habitat conditions required by sage grouse appear not to be met in Duck Creek during much of the growing season. This may explain why, in the past several decades, the number of active leks has declined from three to one in the Duck Creek Allotment (BLM 1979; BLM 2004b). The failure to assess these conditions prevented BLM from adequately determining whether the allotment meets rangeland health standards as they apply to sage grouse.

Management Response to Ecological Assessments

Once the ecological condition of the allotment is assessed and it is determined whether standards are being met, then an evaluation of current management guides the next management decisions. Many of the tools for assessing the influence of management and land use require annual surveys. Plant utilization and stubble height monitoring are two typical annual monitoring activities. Coupled with ecological conditions, these annual monitoring data then should guide changes in grazing use. This section discusses the effectiveness of actions taken by BLM in the study area in response to its assessments of rangeland health.

Grazing Utilization Assessments

Forage utilization is “the percentage of the current year’s herbage production consumed or destroyed by herbivores” (Holechek and others 2004). It is a key guide for determining whether current management is setting grazing use levels to move the allotment towards meeting rangeland health standards. Utilization by livestock and wildlife are key inputs in designing a plan to meet standards. Utilization in the upland areas in Duck Creek is summarized in table 6. Based on paired plot sampling conducted by the authors, utilization in most sites for most years exceeded BLM’s 50 percent utilization standard for upland areas (BLM 2008b). On average, BLM’s utilization data, collected using the key species method, were 31 percent lower than that collected by

the authors. BLM reported utilization was well within the utilization standard of 50 percent. The results of a paired t test comparing BLM’s utilization estimates to the authors’ reported $t = -5.84$ with 17 degrees of freedom. The probability of the null hypothesis (that BLM data equal the authors) is 0.000 percent.

A number of factors explain this discrepancy. The paired plot method used by the authors is quantitative and relies on collection of the grasses and forbs from plots of a standard area, or quadrats. These samples are dried and weighed to determine biomass. The key species method used by BLM is an ocular estimate of the amount of forage removed from plants either by sampling individual plants along a transect or sampling in quadrats. TR 1734-3 states that the use of quadrats is more reliable than the transect, which BLM used in the Duck Creek Allotment. In addition, the key species method requires ungrazed reference plots for comparison. In some years, BLM did not have ungrazed reference plots and thus had to guess what ungrazed conditions would look like. TR 1734-3 requires that observers are trained to estimate utilization and then compare that estimate to clipped and weighed samples. BLM had no records for the utilization training described in TR 1734-3 for the Duck Creek Allotment.

Little research has been conducted to assess whether the key species method accurately represents forage utilization. We can find no studies that validate the method with more quantitative approaches such as the paired plot method. The study usually cited to support the key species method is Heady (1949). Heady (1949) called for utilization estimates to be based on the volume or mass of the plant removed in a “general reconnaissance.” He admitted that these estimates vary widely among individuals or even for one individual between different hours of a day. Holechek et al. (2004) note that the key species method is subjective and its reliability “cannot be readily quantified with standard statistical procedures”. Lastly, BLM (2008b) used many species that are tolerant of grazing as its key species, which leads to management that promotes overutilization and thus decline of the more palatable and less grazing tolerant native bunchgrasses.

A plot of the grass production (kg/ha) in ungrazed upland plots on the Duck Creek Allotment against the grass utilization for the same locations sampled by the authors over five years (52 locations, 1144

samples) is shown in figure 6. This graph shows that when the grass production drops below 200 kg/ha utilization drops to 60 percent or less. Where there is a range of productivity in the uplands, lower utilization may reflect a degraded site with production much lower than potential. This underscores the importance of knowing the actual production at the site where utilization monitoring occurs and of choosing sites that reflect higher production within the allotment. Pinchak et al. (1991) also found that grazing utilization was related to standing crop.

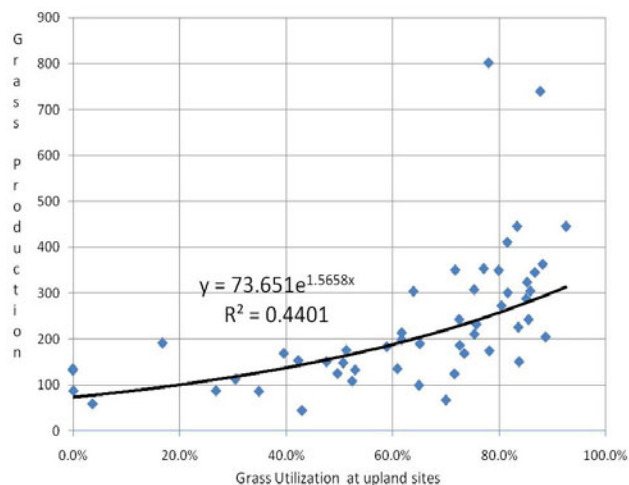


Figure 6. Graph of the correlation of utilization with habitat grass production.

These flaws in the key species method have far reaching consequences. Utilization monitoring provides the key information that BLM uses to change livestock numbers and the duration of grazing. If utilization data are inaccurate or do not represent the desirable forage species, appropriate changes in grazing management are unsupported by this utilization monitoring.

Stubble Height Monitoring

Technical Reference 1734-3 provides BLM with a method to conduct stubble height monitoring (BLM 1996a). A number of assumptions are made when choosing both a key species and a specific height for that species, specifically, that when the stubble height requirement is met: 1) required utilization levels are met; 2) grazing use is moving towards meeting rangeland health standards; and 3) use in the field by agency staff leads to consistent data regardless of the examiner.

For livestock to graze riparian areas without damage, the grazing system must leave adequate residual

stubble height to ensure plant vigor, species diversity, stream bank protection, and sediment capture. To achieve this, minimum herbage stubble height of 10 to 15 cm should be present on all streamside areas at the end of the growing season. For spring grazed pastures, livestock should be removed by July 15, or earlier at lower elevations (Clary and Webster, 1989). Clary and Webster (1989) further recommend that utilization levels should not exceed 40 – 50 percent for summer grazed pastures or 30 percent for fall grazed pastures. Clary and Webster (1989) found that: a 15 cm (six-inch) stubble height corresponded to 24 – 32 percent utilization; four-inch stubble height corresponded to 37 - 44 percent utilization; and a 7 cm (three inch) stubble height corresponded to utilization of 47 – 51 percent. The 15 cm stubble heights should apply to streamside and nearby meadow sites.

Stubble height monitoring has not been closely correlated with ecological habitat conditions. Rather, stubble height is most often tied to the amount of utilization that occurs on the sampled species (McDougald and Platt 1976, BLM 1999c). As a result, stubble height monitoring may be of use in judging the intensity of grazing use but fails to provide a measure of achievement of rangeland health standards (BLM 1997) which are ecologically based.

BLM's most common use of stubble height monitoring is in riparian areas. For the Duck Creek Allotment, BLM used Nebraska sedge (*Carex nebrascensis* Dewey) and Baltic rush (*Juncus balticus*) for stubble height monitoring. Both species persist in degraded riparian conditions in this allotment. Out of the 80 sedge species listed in Hurd et al.'s (1998) "Guide to Intermountain Sedges" only one species, Nebraska sedge, is reported to be tolerant of livestock grazing. Because it is rhizomatous and offers poor forage for grazing animals, Baltic rush is also resistant to grazing (Utah State University Cooperative Extension 2010). Choosing a key species that tolerates grazing means that measuring stubble height will be inadequate for monitoring those species sensitive to and likely to decline with standard BLM grazing use. In the Duck Creek Allotment, the absence of woody riparian plants from most riparian areas and the low diversity of riparian plant species may be accounted for in part by using grazing tolerant species for monitoring, which leads to extremely high utilization levels on riparian vegetation, including willows (Clary and Webster 1989).

Studies of the use of stubble height monitoring in riparian areas have raised a number of concerns. The University of Idaho Stubble Height Study Team (2004) found that the linkage between stubble height data and riparian function has not been adequately researched and thus stubble height is likely inappropriate to use as the only monitoring method for riparian condition. In the Duck Creek Allotment up to 2010, stubble height has been BLM's only annual monitoring method in riparian areas. Other appropriate monitoring methods could include vegetation composition along the green line, stream bank stability, and regeneration of woody species. Burton et al. (2008) developed a riparian assessment method that incorporates multiple quantitative and qualitative indicators of riparian area condition to respond to concerns raised by the use of a single indicator, specifically stubble height.

A summary of residual stubble height data in riparian areas measured by the authors in the Duck Creek allotment is provided in table 7. BLM's 5" (12.7 cm) stubble height objective was never met during five years of monitoring. Readings were generally less than 7.6 cm (3 inches). Most readings were taken in October, a month after the grazing season for cattle ends.

In 2006 through 2010, a rotation grazing system was put in place in the Duck Creek Allotment. Cattle spent typically one month in each of four pastures. BLM predicted that in September regrowth might be expected in pastures that cattle left earlier. However, utilization and stubble height monitoring in riparian areas showed no difference between a pasture that had been rested for up to three months and one where the cattle had most recently grazed. Dry conditions later in the growing season are typical for this climate and this supports research that has shown that for conditions typical for Duck Creek, summer regrowth is minimal (Lile et al 2003).

BLM generally found stubble heights to be greater than the authors' data by 2.3 to 6.4 cm. There are several reasons for this. BLM measured *Carex* and *Juncus* species and reported the average height of the combined species. Inspection of BLM data shows that the Baltic rush generally had stubble heights of about 5 cm greater than the sedge species. In addition, the stems of Baltic rush are so tough that they tend to pull free from the rootstocks when grazed by livestock, especially cattle (Utah State University

Extension 2010). When stubble height monitoring data are collected using the Baltic rush, the only measurable stems are those that remain largely ungrazed. It is not possible to know how many stems have been pulled free. As a result, stubble height monitoring using this species tends to under report grazing use and over report the actual average height of these plants.

The differences between BLM's and the authors' data may also be due to BLM's measurement of stubble heights in areas with hummocks, standing water, or hoof shear depressions, where the vegetation is more protected and grazed less or last. Such areas are technically not along the greenline where stubble height is normally measured. Further, BLM measures heights of plants that have been trampled and are flat against the soil surface. These are likely to have much longer leaf lengths than those that remain standing during the grazing season.

Our livestock census in the Duck Creek Allotment showed 450 cow-calf pairs in 2006 and 304 cow-calf pairs in 2008. Riparian area utilization was not reduced due to a lower number grazing. In 2006 riparian utilization was 87 percent at Six Mile Creek and 94 percent in the south fork of the same creek. In 2008 when fewer cattle were present in the allotment, riparian utilization was 95 percent. The preference of cattle for riparian areas leads to riparian utilization exceeding the standard at both stocking levels when grazed for one month. This is consistent with long-standing research showing that cattle heavily graze riparian areas before seeking upland forage (Hormay and Talbot 1961, Pinchak et al, 1991).

Stocking Levels - Animal Unit Month Redefined

In addition to meeting rangeland health standards, grazing management must also be within the carrying capacity of the allotment (BLM 2006). "(T)he most important of all grazing management decisions, carrying capacity analysis involves spatial analysis of the forage production, the capacity of the area to support livestock grazing, and the amount that can be allocated to livestock" (Holecheck and others 2004). A key factor in this analysis is how much forage a typical sized cow consumes. The animal unit month (AUM) is the basis of permits, stocking rates and fees for grazing public lands. The AUM, however, does not represent current livestock weights and forage consumption.

BLM and the Forest Service have defined an AUM as: "The amount of forage needed to sustain one cow, five sheep, or five goats for a month. A full AUM's fee is charged for each month for adult animals if the grazing animal (1) is weaned, (2) is 6 months old or older when entering public land, or (3) will become 12 months old during the period of use. The term AUM is commonly used in three ways: (1) stocking rate, as in X acres per AUM; (b) forage allocation, as in Y AUMs in allotment A; and (3) utilization, as in Z AUMs consumed a calculated amount of forage" (BLM 2004a).

This definition of an AUM does not account for actual weight and forage consumption of the various animals listed, and it ignores forage consumption by calves and lambs. Clarification and updating of these values are needed so that livestock producers are charged for the actual forage consumed by their animals and the carrying capacity of the land is not exceeded. This would insure that the Federal Land Policy and Management Act (FLPMA) requirement to graze within the carrying capacity of the allotment is met, and that the FLPMA requirement of sustainable use without permanent impairment of productivity is achieved.

Natural Resources Conservation Service (NRCS 2003), in its National Range and Pasture Handbook, defines an animal unit (AU) as one mature cow of approximately 1,000 pounds and a calf as old as 6 months, or their equivalent, then states, "An animal unit month (AUM) is the amount of forage required by an animal unit for one month" (USDA 2003). BLM has typically used 800 lbs/month of forage as the consumption rate for a cow/calf pair. This is 12 kg per day (26 lb/day) and is consistent with a long-standing definition by the Society for Range Management that an animal unit is "one mature (1000 lb.) cow or the equivalent based upon average daily forage consumption of 26 lbs. dry matter per day" (SRM 1974). This was later revised to define an animal unit (AU) as the forage consumption of one standard mature 1,000-pound cow (454 kg), either dry or with calf up to 6 months old and consuming 26 pounds (12 kg) of air-dry forage per day or 800 pounds (363 kg) per month (Ortmann and others 2000).

There are conflicts among these different definitions. First, the use of 26 lbs/day represents oven-dry weight instead of air-dry weight, which is more commonly used in assessing forage production.

NRCS (2003) further defines the actual forage consumption as 26 pounds of oven-dry weight or 30 pounds of air-dry weight per day as "the standard forage demand for a 1,000 pound cow (one animal unit)". This is 2.6 percent of body weight for oven-dry weight and 3 percent of body weight for air-dry weight of forage. As agencies applied these forage needs in their administrative processes, unfortunately the difference between air and oven dried weights got lost. The resulting process further underestimates forage needs for livestock. Note that there is no forage allowance for the calf even though the definition of an animal unit includes a calf. The same is true for lambs, when considering sheep grazing.

Second, these definitions are outdated in terms of the size of today's cattle based on an analysis of USDA market statistics. The University of Nevada Agricultural Experiment Station published a report on cattle production in 1943. This report analyzed 14 years of ranch operation for 11 ranches in northeastern Nevada. At that time, a mature cow was defined as one unit and a branded calf or weaner as $\frac{1}{2}$ unit, for a combined total of 1.5 units per cow/calf pair. Bulls were considered 1.5 units. For the period 1938 – 1940, the average weight of mature cows when they left the range was 435 kg, calves were 173 kg, and bulls were 554 kg. This means that in the 1930s, a cow/calf pair's weight was 608 kg (1340 lbs). The Forest Service, in its Range Analysis Handbook (USDA 1964) provided a detailed summary of forage consumption for cattle and sheep as air-dry amounts. At that time, an Animal Unit was considered as a 1,000-lb cow, while a cow plus 400-lb calf was considered 1.46 animal units. Air dry forage consumption was 24 lb/day (11 kg/day) for the cow and 33 lb/day (15 kg/day) for the cow/calf pair (USDA 1964).

An analysis of USDA market statistics over time reveals significant increases in live weights of cattle (Uresk 2010). In 1964, live weight of mature cattle averaged 456 kg (1,006 lbs) (USDA 1964). In 1978 when the Federal administration implemented the billing formula, the live weight of slaughter cattle averaged 488 kg (USDA 1979). After this point, cattle weight increases were rapid due to selective breeding and the use of hormones and supplements with the USDA reporting average weight for slaughter cattle at 589 kg (1296 lbs) in 2009 (NASS 2010). This is a 100 kg increase over the USDA reported weights in 1978.

Holechek et al. (2004) summarized the weaning weights of calves grazed on various types of rangelands. The data for the period since 1990 produced an average weaning weight of 195 kg within a range of 173 – 216 kg. Ray and others (2004) gave a weaning weight of 218 kg for calves. Using the current market statistics for slaughter cattle of 589 kg and, in the absence of current data use the average weaning weight of 195 kg provided by Holechek et al. (2004), today's estimated average weight of a cow/calf pair during the grazing season is 784 kg.

NRCS estimated that the daily forage consumption for a grazing animal equals 3 percent of its body weight. Thus the combined cow/calf weight of 784 kg consumes 23.7 kg of air-dry forage each day, or 715 kg (1,573 lb) of forage for a month (30.4 days) per AUM. Today's larger weights for cattle make the BLM and SRM definitions of 12 kg/day (26 lb/day) significant underestimate the forage use of today's cattle (Uresk 2010). Based on all of these factors, today's cattle are likely to consume double the amount of forage currently allocated for one AUM. This means that, based on the forage consumption rate alone, current stocking rates should be significantly reduced in the situation where stocking now equals the allotment carrying capacity.

In 2004, BLM made range capacity estimates for the Duck Creek Allotment based on a forage requirement of 2 percent of body weight for a 1,000 pound cow (BLM 2004b). This equates to a requirement for an AUM of approximately 272 kg (600 lb) of forage for each AUM, or 38 percent of the amount consumed by a cow/calf pair today, grossly underestimating the forage demand. By using the same forage

requirement for an AUM that has been in effect since 1961, there is a tendency to overstock an allotment.

Stocking Levels, Carrying Capacity Analysis

Holechek and others (2001) provide a sequence of steps to determine an initial stocking rate for an allotment. This sequence of steps includes determining which lands are capable of supporting livestock grazing: the area must be within two miles of water and have slopes less than 60 percent and produce a minimum amount of forage. When these adjustments are made, most but not all of the Duck Creek Allotment is capable of supporting livestock grazing. The forage available for those lands capable of livestock grazing is determined using reductions for different categories of slope and a reduction in available forage for distances between one and two miles from water. In its 2008 decision (BLM 2008b), BLM did apply these considerations in making a capacity analysis. Based on the updated information that we have assembled, we estimated that the carrying capacity of these lands is actually less than BLM asserts and, thus, the number of livestock that the Duck Creek Allotment might support is also less.

We estimated current forage production in Duck Creek based on the annual production of grasses, since the dominant shrub, sagebrush, and most forbs offer poor forage for cattle. Table 17 presents the dominant forb species identified by BLM and the authors. Species that had annual production of 12 kg/ha or more and were found at a number of sites at that production level are included in table 17. The data indicate that the forbs that dominate the Duck Creek Allotment are not desirable livestock forage species and are not considered in the forage base in this allotment.

Table 17. Palatability of dominate forbs in the Duck Creek Allotment.

Symbol	Common name	Scientific name	Preference
ACNI2	Common yarrow	<i>Achillea millefolium</i>	UUUU
ANMI3	Pussy toes, littleleaf	<i>Antennaria microphylla</i>	NNNN
ASTRA	Vetch, timber milk	<i>Astragalus miser</i>	UDUU
ERCA8	Matted buckwheat	<i>Eriogonum caespitosum</i>	UUUU
LIDAD	Toadflax, dalmatian	<i>Linaria dalmatica</i>	UUUU
PHHO	Hoods (spiny) phlox	<i>Plox hoodii</i>	NNNN
PACA15	Groundsel, wooly	<i>Packera cana</i>	NNNN
SYAS3	Aster, western	<i>Symphyotrichum ascendens</i>	NNNN
ZIPA2	Deathcamas, foothill	<i>Zigadenus paniculatus</i>	TTTT

Cattle grazing preference by quarter of the year: N = not used, D = desirable, P = preferred, T = Toxic, U=undersirable. Species Source: Bureau of Land Management. 2005. Ecological site inventory data at 28 sites in Duck Creek Allotment (species found at 12 kg/ha or more in abundance at several sites). Cattle preference source: Natural Resources Conservation Service. 2005. Ecological site description RO34AY222WY loamy 10-14, animal preferences, quarterly for commonly occurring species. pp 8-9.

Forage production based on current grass production is described in table 5. The range site information comes from the county soil survey (SCS 1982). The authors placed forage production monitoring cages in five of the nine range sites found in the Duck Creek Allotment. The resulting data from our sites represent 98 percent of the area of the allotment. Multiple sample sites were located in most range sites, and the amounts of forage found at these sites were averaged together. Grass production for 2005 was not used because precipitation was above average; other years had average precipitation.

Cattle have access to herbaceous plants that are primarily located in the shrub interspace area. This carrying capacity analysis assumes that grass growing under shrubs is not available as a forage source to cattle. Based on the canopy cover survey the authors conducted, 67 percent of the allotment is interspace area between shrubs (table 9). The available area for forage was determined by multiplying the area in a range site with a slope less than 50 percent (BLM slope criterion) by this interspace factor of 67 percent. The total production for a Duck Creek range site is the result of multiplying the available area times the grass production of that range site.

If we make the standard assumption often used by BLM—that one AUM uses 272 kg (600 lbs)/month, forage under shrubs and in interspaces can be grazed and 50 percent of palatable forage is *allocated to livestock*—we find that the Duck Creek Allotment will support 2,479 AUMS and produce 1,348,681 kg for the allotment based on the authors' forage production data. The Duck Creek Allotment is currently managed to allow 3,320 AUMs of grazing use.

Using data that reflect the weight of today's cows and the light utilization required for impaired lands in this region, one AUM requires 706 kg (1,556 lb)/month and 30 percent of the grass production would be allocated to livestock (Holechek et al. 2004). Based on these assumptions using the authors' forage production data, the current carrying capacity for the Duck Creek Allotment is 581 AUMs or 18 percent of what is now permitted to graze.

A more detailed analysis of forage capacity of this allotment is likely to lead to the conclusion that this

allotment will support even fewer livestock. One key consideration, not incorporated in the capacity analysis in the previous paragraph, is the erodibility of soils. Highly erodible soils are unlikely to sustain domestic grazing under traditional grazing practices (USDA 2003). Erodible areas that cannot sustain livestock grazing because of biophysical limitations are classified as not capable or suitable for livestock grazing (USDA 2004, BLM 1979). Areas identified with high to very high potential for erodibility should be classified as unsuitable for livestock grazing and not included in carrying capacity analysis. Based on the Rich County soil survey (SCS 1982), almost half of the allotment has soils with high or very high erodibility. Reducing the amount of land capable and suitable for grazing will further reduce the capacity of the allotment.

The authors argue that capacity analysis should also account for the amount of herbaceous plant production needed to support wildlife. Except for major game ungulates, range capacity fails to account for this key need. The authors reviewed the forage demands for common mammals that occupy the Duck Creek Allotment, table 18 (Catlin et al. 2003), and found that about 225 kg/ha per year should be allocated to mammalian herbivores in sage steppe. To calculate this allocation, we selected three primary herbivores (or in the case of folivorous/omnivorous rodents, a guild) that fairly represent the mammalian herbivores present in sage steppe: mule deer, jackrabbits, and rodents. More study is needed to validate the estimates in table 18 for this specific locale. Based on wildlife needs in the Duck Creek Allotment, it is probable that 5-30 percent of the annual plant production is needed to support wildlife when making a range capacity analysis. When wildlife forage needs are included into the range capacity analysis, the carrying capacity for livestock will be further reduced.

BLM argues that the current stocking number is well within the forage production capacity of this allotment (BLM 2008b). We argue--based on the best available information concerning forage production, livestock consumption, habitat and wildlife needs--that the livestock number that can be supported in this allotment is substantially lower than what is now permitted. Grazing at levels above the allotment's carrying capacity leads to high utilization levels, shift in the plant community away from potential, and increased degradation of riparian areas.

Table 18. Kg/ha/year of forage (grass and forb) biomass necessary to support typical mammal herbivore populations in arid Utah.

Species	Density (Individuals per hectare)	Average total forage per individual (kg./day/individual)	Herbaceous forage in diet (percent)	Herbaceous forage per individual (kg./day per individual)	Herb. forage per population per day (kg/ha/day)	Herb forage per population per year (kg/ha/year)
Deer	0.11	1.58	22.40%	0.325	0.035	12.73
<i>Deer Lit Citations</i>	<i>1*,34,42,43,45</i>	<i>2,9,11,12,27,29,37</i>	<i>4,10,14,27, 29,29</i>			
Jackrabbits	2.01	0.13	74.70%	0.097	0.199	72.66
<i>Jackrabbit Lit Citations</i>	<i>5,6,8,20,22, 29,40</i>	<i>4,15,15,22,29, 32,33,36</i>	<i>3,21,23,24, 32</i>			
Rodents	16.3	0.056	43%	0.024	0.39	142.3
<i>Rodent Lit Citations</i>	<i>16,17,18,19,25, 26,28,38, 39,46,47</i>	<i>29,30,31,35</i>	<i>38,41,44,48</i>			

Total Herbaceous Forage Allocation For Mammalian Herbivores =**227.6 kg/ha/yr**

*References are as follows: 1. Chapman & Feldhamer 1982, 2. Demaras & Krausman 2000, 3. Fagerstone et al. 1980, 4. Krausman 1996, 5. Daniel et al. 1993, 6. Johnson & Anderson 1984, 7. Kufeld 1973, 8. Anderson and Shumar 1986, 9. Smith 1953, 10. Bueker et al. 1972, 11. Aldredge et al. 1974, 12. Smith 1952, 13. Hobbs et al. 1982, 14. Hansen & Clark 1977, 15. Currie and Goodwin 1966, 16. Fautin 1946, 17. Grant et al. 1982, 18. Nelson & Legee 1982, 19. Grant & Birney 1979, 20. Norris 1950, 21. Fagerstone et al. 1980, 22. Arnold 1942, 23. Alipayo 1991, 24. Wansi 1989, 25. WRSOC 1983, 26. Hanley & Page 1981, 27. Urness 1981, 28. Rosenstock 1996, 29. Stoddart et al. 1955, 30. Golley 1960, 31. Kuford 1958, 32. Hoffmeister 1986, 33. McAdoo & Young 1990, 34. UDWR 2003, 35. Detling, in prep, 36. Vorhies 1933, 37. Jensen 1984, 38. Goodwin & Hungerford 1979, 39. Shepard 1972, 40. Stoddart 1938, 41. Black & Frischknecht 1971, 42. Horejsi & Smith 1983, 43. Clegg 1994, 44. BLM 1998, 45. AGFD 2003, 46. West 1983a, 47. West 1983b, 48. Alston 2002.

Drought Management

BLM's drought management policy includes consideration of the U.S. Drought Monitor forecasts, and early assessment of on-the-ground conditions to determine management actions, including possible reductions in grazing to accommodate drought (BLM 2003). The U.S. Drought Monitor has provided assessments of drought since 1999 and shows that for the period 2000 to 2009, drought was experienced on the Duck Creek Allotment 7 out of these 10 years. Except for the above average precipitation year in 2005 when BLM conducted surveys, most years have average or below average precipitation (Fig. 4). BLM sends out drought notices periodically, but no evidence of destocking has been found in billing records or actual use reports. Some notices were sent out near the end of the grazing season, too late for meaningful action, even though drought had been identified months earlier.

Holechek et al. (2004) recognize that livestock stocking rates should be reduced in accordance with forage capacity. Forage production varies with precipitation and can range widely between dry and

favorable years (SCS 1982). After drought, the ability of forage plants to recover is directly related to the standing crop levels maintained during the dry period (Holechek et al. 1999b). It has long been recognized that dry years (below average precipitation years) occur about 50 percent of the time (Hutchings and Stewart 1953). These authors suggested that 25 – 30 percent use during average precipitation years of all forage species by livestock is proper. They recommended this level because routinely stocking at capacity will result in overgrazing in half the years and necessitate heavy use of supplemental feed. Even with this system, they recognized that complete destocking would be needed early into, during, and after drought (Thurow and Taylor 1999).

Drought management should reflect the need to restore degraded habitat prior to drought. The Duck Creek Allotment contains degraded native plant communities, soils exposed to accelerated erosion, and degraded riparian systems. These conditions have been exacerbated by BLM management during drought and dry years. BLM has not adequately monitored and managed the public lands for their

potential or sustained use. The result is that productivity has been impaired and will be impaired permanently unless management changes are based on science and objective, quantitative assessments.

Discussion of Grazing Practices

Research over the past several decades provides solutions to the livestock induced problems on the Duck Creek Allotment and millions of acres of public lands across the West. Drought has become a persistent condition on the Duck Creek Allotment, and management should accommodate these conditions as they become normal with climate change. Failure to adjust stocking rates within current capacity and reduce stocking to account for lower forage production in dry or drought years has potentially serious negative ecological impacts.

High stocking rates have led to high utilization on the Duck Creek Allotment and to shifts in the native plant community to less desirable species and lowered productivity. The substantial decline of a keystone native bunchgrass, bluebunch wheatgrass, exemplifies the cost of over-utilization. BLM has consistently allowed heavy use (50 percent or more) to occur on the allotment's uplands and 90 percent in riparian areas. Research has shown that utilization levels of 30 percent or lower improve productivity. Holechek et al. (1999, 2004) have found that during drought moderately stocked pastures produce 20 percent more forage than heavily stocked pastures, and lightly stocked pastures produce 49 percent more forage than heavily stocked pastures and 24 percent more forage than moderately stocked pastures.

In 2005, the north half of the Duck Creek Allotment was rested. Monitoring after this rest period showed no measurable herbaceous plant community improvement. From 2006 to 2009, a four pasture deferred system of grazing was followed. Utilization in riparian areas continued to exceed 90 percent and regrowth was not evident in any of the pastures. Deferred grazing systems such as BLM is implementing on the Duck Creek Allotment have shown no advantage over season-long grazing (Briske and others 2008). Stocking rate adjustments have proven effective in increasing forage production if utilization does not exceed 30 percent (Briske et al. 2008, Clary and Webster 1989, Eckert and others 1986, 1987, Holechek et al. 1998, Holechek et al. 2000, Van Poolen et al. 1979).

Except for rest for half of the allotment in 2005, rest has not been provided in the Duck Creek Allotment for other years and pastures. Lack of a full growing season for rest and high utilization may explain the low vigor of the native bunchgrass communities (Anderson 1991, Hormay and Mueggler 1975, Mueggler 1975, Talbot 1961). In studies of long-term rest at Idaho National Engineering Laboratory, the recovery rate of grasses in sagebrush communities was slow but real, progressing from 0.28 percent to 5.8 percent ground cover over 25 years (Anderson and Holte, 1981), and non-natives such as cheatgrass had an inverse relationship to native perennial grasses (Anderson and Inouye 2001).

CONCLUSIONS

On western rangelands, livestock grazing as has been traditionally practiced has significantly reduced wildlife habitat resilience (Belsky and others 1999, Bruan 2006, Fleischner 1994, Fleischner 2010, Jones 2000,). This paper presents a more comprehensive analysis in order to understand the relationship between ecological theory, land management policy, habitat management standards, agency ecological assessment methods, and how these are practiced in the field. As the authors' analysis shows, specific on the ground data gathering was critical in order to link field application with policy and theory.

Secretary of the Interior Salazar has committed his agency to "three new functions: renewable energy production, carbon capture and storage, and climate adaptation" (Salazar 2009). Carbon storage and climate adaptation are both relevant to range management. Through agency-promoted ecosystem restoration, storage of organic carbon in soils and plants could increase according to Salazar. About 13 percent of soil organic carbon is stored in shrublands (Sundquist et al. 2009). We do not know the amount of increase in stored organic carbon that we might see if those lands reached their ecological potential. The ecological assessment methods reviewed in this paper typically don't assess the amount of carbon stored in soils. Correction of this shortcoming is not planned at this time. Failure in the past to accurately assess carbon storage and other ecological indicators is also not recognized as a research need by the federal government (U.S. Global Change Research Program, 2009). However, the need for change in range management has not been articulated in agency responses to climate change up to this point.

The authors argue that promoting resilient habitat is a key part of the adaptation needed to reduce the impacts of climate change in the West. As is detailed in this paper, BLM habitat assessment methods by design often under report habitat that has significantly departed from its ecological potential, and thus has lost its resilience. Based on the ecological assessments that BLM has conducted in Utah, only 1 percent of the assessed allotments require changes in range management in order to meet rangeland health standards. Our research on the Duck Creek Allotment suggests that rangelands have experienced a significant loss of resilience, and that this has not been captured fully by agency monitoring and analysis.

In order to understand what might be the cause of the disconnect between agency ecological assessments and ecosystem condition, several analyses were required. Each element of the research presented here provides needed insight into what causes agency assessments to conflict with measured ecological condition. Part of the problem can be explained in the design of agency ecological assessment methods. A review of BLM policies and assessment methods shows that key ecological indicators are missing from BLM's ecological assessment methods. BLM's rangeland health standards cover many of the required ecological factors, but they do not incorporate these indicators at the spatial and temporal scales needed.

BLM has preferred to use qualitative ecological assessment methods that, judged by the authors' data, fail to meet federal requirements for assessing compliance with BLM's standards. As our critique of these assessment methods shows, independent review and validation of agency assessment methods is seriously needed. The use of these methods in the field, as demonstrated in this study, has under reported ecological problems.

The consequences of BLM's failure to adequately assess habitat conditions on the Duck Creek Allotment are significant. BLM's analysis failed to identify the significant loss of the key dominant bunchgrass community, the loss of overall productivity, the excessive amount of bare ground in most ecological sites, a shift in the plant community towards lower biodiversity dominated by grazing tolerant plants, the almost complete loss of woody riparian plants, and, likely, a reduction in wildlife

populations. As a result, today Duck Creek has no ducks.

Likewise, BLM's trend, utilization and stubble height monitoring data are not consistent with the authors' data. BLM's qualitative ocular methods consistently reported utilization levels over 31 percent less than levels determined by quantitative methods. Grazing utilization in upland areas was well above the required management standard of 50 percent and was over 90 percent in riparian areas. BLM claims to rely on its utilization and stubble height data to seasonally adjust the amount of grazing each year. Based on the Duck Creek Allotment data presented in this study, the methods BLM used consistently under reported utilization and are inappropriate for making accurate stocking level decisions.

Because of this problem with BLM monitoring, carrying capacity analysis is needed. Unfortunately, BLM has rarely conducted range capacity analyses in the past 25 years (Robinson 2008). To be consistent with today's conditions and the agency's ecological management direction, range capacity analyses needs to be updated West wide. Forage demand by livestock has changed over time and stocking decisions made by BLM fail to address this change. The forage needs of today's livestock are a key input in any carrying capacity analysis. The increase of the weight of cattle today indicates that today's cows consume more than BLM currently allocates. And, the ecological needs of wildlife should also be incorporated into range capacity analysis, with special attention to ecological restoration. This study estimated, based on field data and current recommendations for grazing practices, that BLM had significantly overstocked the Duck Creek Allotment.

Drought will become the norm in the future. Preparation for potential drought conditions requires actions prior to drought to reduce land use impacts, as well as a recovery period after a drought. Based on BLM's record in the Duck Creek Allotment, response to droughts has been minimal and too late to be effective. Rest or stocking reductions of livestock needed for drought management or post drought recovery have not occurred. In 2006, Congress established the National Integrated Drought Information System (NIDIS Act), which incorporated existing and new drought data and prediction analysis into a coordinated program. Based on BLM's records for the Duck Creek Allotment, agency use of these

data to predict and respond to drought has not occurred.

BLM did recognize that new management was needed to address problems in some riparian areas in the Duck Creek Allotment. In the first phase of BLM's revised management scheme, the allotment was divided into four pastures, with grazing occurring in each pasture each year for one month on a rotating schedule. Our study for this allotment has field data prior to and for several years during this first phase. Based on comparing pre and post deferment data, conditions in this allotment show almost no improvement in riparian and upland areas. While the number of livestock grazed has often been less than the permitted number, the data show continued degradation. Phase two of the revised management scheme recently placed upland water troughs in these pastures and data are now being collected to identify any resulting changes. It is too early to evaluate this second phase.

Holling and Meffe (1996) provide a model that helps explain the characteristics on the ground of BLM's current range program in the Duck Creek Allotment. Holling (1995) argues that when socioeconomic goals dominate "any attempt to manage ecological variables (e.g. fish, trees, water, cattle) this inexorably leads to less resilient ecosystems, more rigid management institutions, and societies more dependent on resource extraction." Gunderson & Holling (2002) label this as a pathology of resource and ecosystem management.

The refusal by BLM to implement proven solutions to overgrazing illustrates Gunderson and Holling's concept of pathological management. Rest, both growing season long and over many years, is normally required for habitat recovery (Kowalenko and Romo 1996, Thurow and Taylor 1999). Further, once recovery has occurred, stocking levels must be set to ensure that habitat remains at its ecological potential. Changes in grazing systems (deferred, rotational, short duration rotation, rest rotation, etc.) alone do not address the problems caused by overstocking (Briske et al. 2008).

The extent to which habitat condition departs from ecological potential is a significant factor influencing the severity of impacts from drought (Bahre and Shelton 1993). The examples that compare impaired

streams and sagebrush habitat with nearby sites that are near ecological potential demonstrate the enormous importance of resilient habitat to ecosystem support in a time of drought. Habitat at its ecological potential is likely to be impacted less from climate change than predicted (West and others 2009). Methods are available for assessing habitat resilience; but as our Duck Creek Allotment study has shown, BLM's current range management program falls far short of identifying loss of habitat resilience and taking action to correct that loss.

We see the new direction of Interior as an opportunity to promote resilient rangelands as a key part of our response to climate change. As this paper shows, significant change in BLM is needed in order to assess the health of ecosystems and manage in deference to habitat health. History has shown that BLM is unlikely to address this need solely through internal means. Engagement of the scientific community is required. Ronald Reagan (1987) advocated a policy of "trust but verify." Clearly the concept of external verification applies to range management as well as to foreign policy.

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Appendix A. BLM Upland Rangeland Health assessment results for Duck Creek allotment, 2005.

Site ID	1 rill	2 water flow	3 pedestals	4 bare ground	5 gullies	6 wind scour	7 litter move	8 resist erosion	9 soil loss	10 plant comp	11 compaction	12 function groups	13 plant mortality	14 litter amount	15 Annual Prod	16 invasive	17 repro perennials	Assessment Results	Similarity Index (Normal Precip.)
DC-1	5	5	5	5	5	5	5	5	5	4	5	4	5	5	5	2	5	F	66%
DC-2	4	2	4	5	3	5	4	3	5	3	4	2	4	3	3	2	4	FAR	54%
DC-3	5	5	5	5	5	5	5	5	5	5	5		5	5	5	5	5	F	56%
DC-4	5	5	5	5	5	5	4	5	5	5	5	5	5	5	5	5	5	F	63%
DC-5	5	4	5	5	5	5	5	5	5	5	5	4	5	5	5	5	5	F	48%
DC-6	5	5	5	5	5	5	5	5	5	4	5	5	5	5	5	2	5	F	68%
DC-7	5	5	5	5	5	5	5	5	5	4	5	4	5	5	5	4	5	F	66%
DC-8	5	5	5	5	5	5	5	5	5	4	5	4	5	5	5	4	5	F	68%
DC-9	5	5	5	5	5	5	5	5	5	4	5	4	5	5	5	3	5	F	68%
DC-10	5	5	5	5	5	5	5	5	5	4	5	4	5	5	5	5	5	F	68%
DC-11	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	F	56%
DC-12	5	5	5	5	5	5	5	5	5	4	5	5	5	5	5	4	5	F	78%
DC-13	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	F	73%
DC-14	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	F	85%
DC-15	5	5	5	5	5	5	5	5	5	5	5	5	4	5	5	5	5	F	59%
CD-16	5	5	5	5	5	5	5	5	5	5	5	5	4	5	5	5	5	F	59%
DC-17	5	5	5	5	5	5	5	5	5	5	5	5	4	5	5	5	5	F	53%
DC-18	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	F	53%
DC-19	5	5	5	5	5	5	4	5	5	5	5	4	5	4	5	3	5	F	49%
DC-20	4	5	5	5	4	5	5	5	5	5	5	5	5	5	5	5	5	F	20%
DC-21	5	5	5	5	4	5	5	5	5	5	5	5	5	5	5	5	5	F	58%
DC-22	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	F	53%
DC-23	5	5	5	5	5	5	4	5	5	5	5	4	5	4	5	3	5	F	72%
DC-24	5	5	5	5	5	5	5	5	5	4	5	4	5	5	5	5	5	F	61%
DC-25	5	4	4	5	5	5	5	4	5	4	4	3	4	4	3	3	4	FAR	40%
DC-26	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	4	5	F	45%
DC-27	4	3	4	5	3	5	4	3	5	3	5	4	4	4	4	4	5	FAR	50%
DC-28	5	5	5	5	5	5	5	5	5	5		5	5	5	5	4	5	F	64%

Appendix B. Results of BLM lotic (stream) PFC assessments, Duck Creek Allotment, 2005.

Stream Segment	Stream Name	1 floodplain inundation	2 Beaver	3 sinuosity, width, gradient	4 riparian area widening	5 uplands not degrading stream	6 diverse age class	7 diverse composition	8 soil moisture	9 roots withstand floods	10 plants high vigor	11 plants protect banks	12 large woody debris	13 channel dissipates flood	14 point bars reveg.	15 stream lateral migration	16 stream vertically stable	17 sediment/flow in balance.	Function
DC-R1A	Duck Creek	N	NA	YN	N	Y	Y	Y	Y	Y	Y	YN	N	Y	Y	Y	Y	Y	FAR
C- R1B	Duck Creek	Y	NA	Y	Y	Y	Y	Y	Y	Y	Y	YN	NA	Y	Y	Y	Y	Y	FAR
DC-R15	Six Mile Creek	Y	NA	YN	YN	Y	Y	Y	Y	YN	YN	YN	NA	YN	Y	Y	Y	Y	FAR
DC- R21	Six Mile Creek	YN	NA	Y	Y	Y	YN	YN	Y	YN	Y	YN	NA	YN	YN	Y	Y	Y	PFC
DC-R21B	Six Mile Creek	N	NA	YN	Y	Y	YN	YN	YN	YN	YN	YN	NA	YN	N	YN	Y	Y	FAR
DC-R29	Sage Creek, Sec 18 SE/SW	Y	NA	Y	YN	Y	Y	Y	Y	YN	Y	Y	NA	Y	YN	YN	Y	Y	FAR
DC-R31	Reach A, NF 6 mi	Y	NA	YN	Y	Y	Y	Y	Y	Y	Y	Y	NA	Y	Y	Y	Y	Y	PFC

YN indicates that the answer was between yes and no.

Appendix C. Results of BLM lentic (Spring) PFC assessments, Duck Creek Allotment, 2005.

Site	Stream Name	1 Wetland saturated frequently	2 fluctuating water level	3 wetland area at potential	4 uplands not degradating wetlands	5 water quality supports plants	6 water flow not disturbed	7 structure not inhibiting flow	8 diverse plant age class	9 diverse plant composition	10 Maintain soil moisture	11 root mass prevents erosion	12 plants have high vigor	13 Plant cover protects soils	14 no frost heaving	15 favorable microsite condition	16 chemicals not affect plants	17 soil soil saturation good	18 impermeable geologic layer	19 water sediment balance	20 shores endure waves	Functional Rating
DC-R2	Duck Creek	Y	Y	Y	Y	Y	Y	na	Y	Y	Y	Y	Y	Y	Y	n	Y	Y	na	Y	na	PFC
DC-R3	Duck Creek	Y	Y	n	Y	Y	Y	na	Y	Y	Y	Y	Y	Y	Y	na	Y	Y	na	Y	na	PFC
DC-R4	(see R3)	Y	Y	n	Y	Y	Y	na	Y	Y	Y	Y	Y	Y	Y	na	Y	Y	na	Y	na	PFC
DC-R5	DC spring 4	Y	Y	n	Y	Y	Y	na	Y	Y	Y	Y	Y	Y	Y	na	Y	Y	na	Y	na	PFC
DC-R6	Spring 5	Y	Y	n	Y	Y	n	na	Y	Y	Y	Y	Y	Y	Y	na	Y	Y	na	Y	na	FAR
DC-R7	Rich Spring	Y	Y	n	Y	Y	Y	na	Y	Y	Y	Y	Y	Y	Y	na	Y	Y	na	Y	na	PFC
DC-R8	Duck Creek	Y	Y	n	Y	Y	Y	na	Y	Y	Y	Y	Y	Y	Y	na	Y	Y	na	Y	na	PFC
DC-R9	(See R8)	Y	Y	n	Y	Y	Y	na	Y	Y	Y	Y	Y	Y	Y	na	Y	Y	na	Y	na	PFC
DC-R10	(See R8)	Y	Y	n	Y	Y	Y	na	Y	Y	Y	Y	Y	Y	Y	na	Y	Y	na	Y	na	PFC
DC-R11	(See R8)	Y	Y	n	Y	Y	Y	na	Y	Y	Y	Y	Y	Y	Y	na	Y	Y	na	Y	na	PFC
DC-R12	DC spring 14	Y	Y	Y	Y	Y	Y	na	Y	Y	Y	Y	Y	Y	Y	na	Y	Y	na	Y	na	PFC
DC-R13	Red Spring (by highway)	n	n	n	Y	na	n	na	Y	Y	Y	Y	Y	Y	Y	na	Y	n	na	Y	na	FAR
DC-R14	Hidden Spring	n	Y	n	Y	n	n	na	n	n	n	n	n	Y	Y	na	Y	n	na	Y	na	NF
DC-R16	Up stringer meadow	Y	Y	n	Y	Y	Y	na	Y	Y	Y	Y	Y	Y	Y	na	Y	Y	na	Y	na	FAR
DC-R17	(same as R16)	Y	Y	n	Y	Y	Y	na	Y	Y	Y	Y	Y	Y	Y	na	Y	Y	na	Y	na	FAR
DC-R18	North Spring	Y	Y	Y	Y	Y	Y	na	Y	Y	Y	Y	Y	Y	Y	na	Y	Y	na	Y	na	FAR
DC-R19	Barrel Spring	Y	Y	Y	Y	Y	Y	na	Y	Y	Y	Y	Y	Y	Y	na	Y	Y	na	Y	na	FAR
DC-R20	Sec 33 spring, S.F 6 mile	Y	Y	Y	Y	Y	Y	na	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	na	Y	na	PFC
DC-R22	Spring 1, N. 6 Mile	Y	Y	Y	Y	Y	Y	na	Y	Y	Y	Y	Y	Y	Y	na	Y	Y	na	Y	na	PFC
DC-R23	Rabbit Spring																					
DC-R24	Spring 3, 6 Mile	yn	yn	n	Y	Y	Y	na	Y	Y	Y	Y	Y	Y	Y	na	Y	Y	na	Y	na	FAR
DC-R25	Meadow Sec 25 E1/2																					
DC-R26	Spring 2, Sec 25 NW1/4	Y	Y	Y	Y	Y	Y	na	Y	Y	Y	Y	Y	Y	Y	na	Y	Y	na	Y	na	PFC
CD-R27	McKinnon Spring																					
DC-R28	Sec 31 spring	Y	Y	n	Y	Y	n	na	n	n	Y	Y	Y	Y	Y	na	Y	Y	na	Y	na	NF
DC-R30	Sage Creek spring	Y	Y	Y	Y	Y	Y	na	Y	Y	Y	Y	Y	Y	Y	na	Y	Y	Y	Y	na	PFC

na means that the indicator was not applicable for that site. yn indicates that the answer is in part both yes and no for the indicator

A GIS Ordination Approach to Model Distribution of Shrub Species in Northern Utah

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ABSTRACT

*Anthropogenic and natural disturbances represent a serious threat to natural ecosystems dominated by big sagebrush (*Artemisia tridentata*). Conservation efforts aim to restore original species composition and prevent the invasion of undesirable species. In order to restore the historic plant communities, we need a clear understanding of how species compositions are distributed along environmental gradients. Species ordination is a process of placing plant species along environmental gradients. This study was conducted in Rich County, Utah, where substantial changes in species composition have been documented in recent years. Field data, literature review, multivariate analyzes, GIS and remote sensing techniques, and expert knowledge were used to define environmental variables and their respective suitability ranges of where shrub species may occur along this area. Ordination and CART-statistical analyzes were used to estimate and predict suitability of shrub species along environmental gradients. GIS procedures were used to spatially predict species distribution. Field data and the Southwest Regional Gap Analysis Project data provided useful information to build the model and 20 percent of field data was withheld to cross-validate the findings. Final results showed that the shrub species distribution in the rangelands of Northern Utah, specifically Rich County, might be driven by precipitation and temperature gradients -influenced greatly by elevation. Slope contributing area, NDVI, and solar radiation were statistically significant factors explaining shrub distribution. To our perception, soil moisture availability might be the most explanatory variable behind these findings. In the model validation, the Kappa coefficient was $K = 61.3$ percent and the overall model accuracy was 74 percent. The location of species distribution areas, in the final map, can be useful to managers in order to define where resources might be allocated to preserve and restore these native rangeland ecosystems.*

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RESUMEN

*Perturbaciones naturales y antropogénicas representan una seria amenaza para los ecosistemas naturales dominados por sagebrush (*Artemisia tridentata*). Los esfuerzos conservacionistas se enfocan en restaurar la composición original de las especies y prevenir la invasión de especies indeseables. Para poder prevenir o restaurar las comunidades vegetales, necesitamos un claro entendimiento de como la composición de especies esta distribuidas a lo largo de gradientes ambientales. La ordination de especies es un proceso de colocar las diferentes especies dentro de un rango de variables ambientales. Este estudio fue conducido en el Condado de Rich, estado de Utah, USA, donde cambios sustanciales en la composición de especies han sido reportados en los últimos años. Datos de campo, revisión de literatura, análisis multivariados, técnicas SIG y de teledetección, así como también el conocimiento de expertos en la materia, fueron utilizados para definir los rangos de variables ambientales sobre los cuales las especies estudiadas de arbustos se localizan. Análisis de regresión usando técnicas de ordination y árboles de decisiones, fueron utilizados para predecir las variables ambientales y sus respectivos rangos, donde estas especies podrían habitar. Datos de campo y resultados del proyecto Southwest Regional Gap Analysis proveyeron de información útil para construir el modelo y 20 percent de las muestras de campo fueron retenidas para validar los resultados. Los resultados finales muestran que la distribución de especies arbustivas en el norte de Utah, específicamente en el Condado de Rich, pueden estar gobernadas por gradientes de precipitación, temperatura -ambas variables influenciadas por la altitud-. El área de la pendiente tributaria, el Índice Normalizado de Diferenciación de la Vegetación (NDVI, por sus siglas en inglés) y la radiación solar, también resultaron estadísticamente significativos como variables predictoras. De acuerdo a nuestra percepción, la disponibilidad de humedad en el suelo podría ser la variable oculta detrás de las otras variables. En la validación del modelo, el Coeficiente Kappa fue de $K = 61.3$ percent y la precisión global del modelo resultó =74 por ciento. La localización de las especies en el mapa final, puede ser de gran ayuda para las agencias de gobierno para decidir donde los esfuerzos de restauración podrían concentrarse para proteger y preservar estos importantes ecosistemas nativos.*

INTRODUCTION

Shrub ecosystems occupy large areas in the western U.S. and have long provided society with grazing opportunities, water, wildlife habitat and recreational values. Nearly 45 million hectares in the western U.S. are dominated by sagebrush ecosystems (*Artemisia* spp.) (West 1999). In recent decades, their abundance and ecological condition has declined in reaction to natural and anthropogenic processes (Wisdom et al. 2005a). Documented examples of such processes include the invasion of non-native, colonizing herbaceous species (i.e. Cheatgrass (*Bromus tectorum*) mainly on the warmer and drier low altitudes, and the encroachment of woodlands, such as Pinyon-Juniper, in the cooler and wetter and higher altitudes (Suring et al. 2005, Wisdom et al. 2005b). Land management concerns include the loss of prime agricultural land, urban growth and encroachment, loss of prime habitat, regrowth of native vegetation following wildland fire events, erosion, rangeland and forest health changes due to management prescriptions, and the distribution and expansion of wide-ranging noxious weeds (Holechek et al. 1989). Both human and natural perturbations have a significant impact on these sagebrush ecosystems.

Species ordination may assist in restoring these natural ecosystems to their original species distribution. Species ordination is the process of placing species along one or more environmental gradients or to abstract axes that may represent such gradients (Austin 1985). The objective of ordination is to locate patterns of species composition along gradients. Intents for species ordination and classification started at the beginning of last century. In 1930, Ramensky began to use informal ordination techniques for vegetation. Such informal and largely subjective methods became widespread in the early 1950's (Austin 1985). Whitaker introduced the unimodal model concept, in which species abundance was a function of a position along a single gradient (Whittaker and Niering 1965). Today, ordination may be seen as an exploratory data-analysis technique that identifies pattern, such as trends, clusters or outliers, using a multivariate set of data.

Decision-tree classifiers are well appropriated for land cover mapping, especially when considering multiple environmental explanatory variables spatially distributed over an area (Vayssières and Plant 1998).

First, as a non-parametric classifier, decision trees require no previous assumptions of normality, which is useful as many land-cover classes and when environmental features do not show a normal distribution. Second, decision trees accept a variety of measurement scales in addition to categorical variables, which may be the case while using ancillary data (De'Ath 2002). Traditional parametric classifiers have difficulty dealing with differences in spatial and ancillary measurement scales. Decision-tree classifiers have demonstrated improved accuracies over the use of traditional classifiers (Dixit and Geevan 2002). Finally, decision tree software is readily available, computationally efficient, and by using a hierarchical approach to define decision rules, is relatively user-friendly to a variety of users. (Lowry et al. 2005, 2007).

To our knowledge, linking multivariate ordination studies and GIS analysis is a relatively novel task. Few studies report the use of spatially explicit ordination data to place areas of species occurrence in maps (Merzenich and Frid 2003). Some other studies mention the use of GIS data to determine the values of environmental variables used in the ordination process. The purpose of this study was to spatially predict the occurrence of seven sagebrush shrub types in the rangelands of Rich County, Utah using a GIS predictive model.

METHODS

Study Area

The study area was located in Rich County, Utah. The rangelands of Rich County in Northern Utah are characterized by having vegetation dominated by sagebrush (*Artemisia* spp.) communities associated with native and introduced grasses (Stoddard 1940). Rich County is predominantly composed of salt desert scrub, big sagebrush-steppe and shrublands, as well as pinion-juniper ecosystems (Washington-Allen et al. 2004, 2006). Rich County is best characterized as a higher elevation big sagebrush-steppe/shrubland environment ranging from the pinion-juniper ecosystems to sub-alpine forests and meadows. Our work focused on the big sagebrush-steppe shrublands and pinion-juniper ecosystems. Both study areas have suffered changes due in historical disturbance regimes ranging from grazing, burning, drought, and flooding events. These areas have been under commercial agriculture and grazing for years.

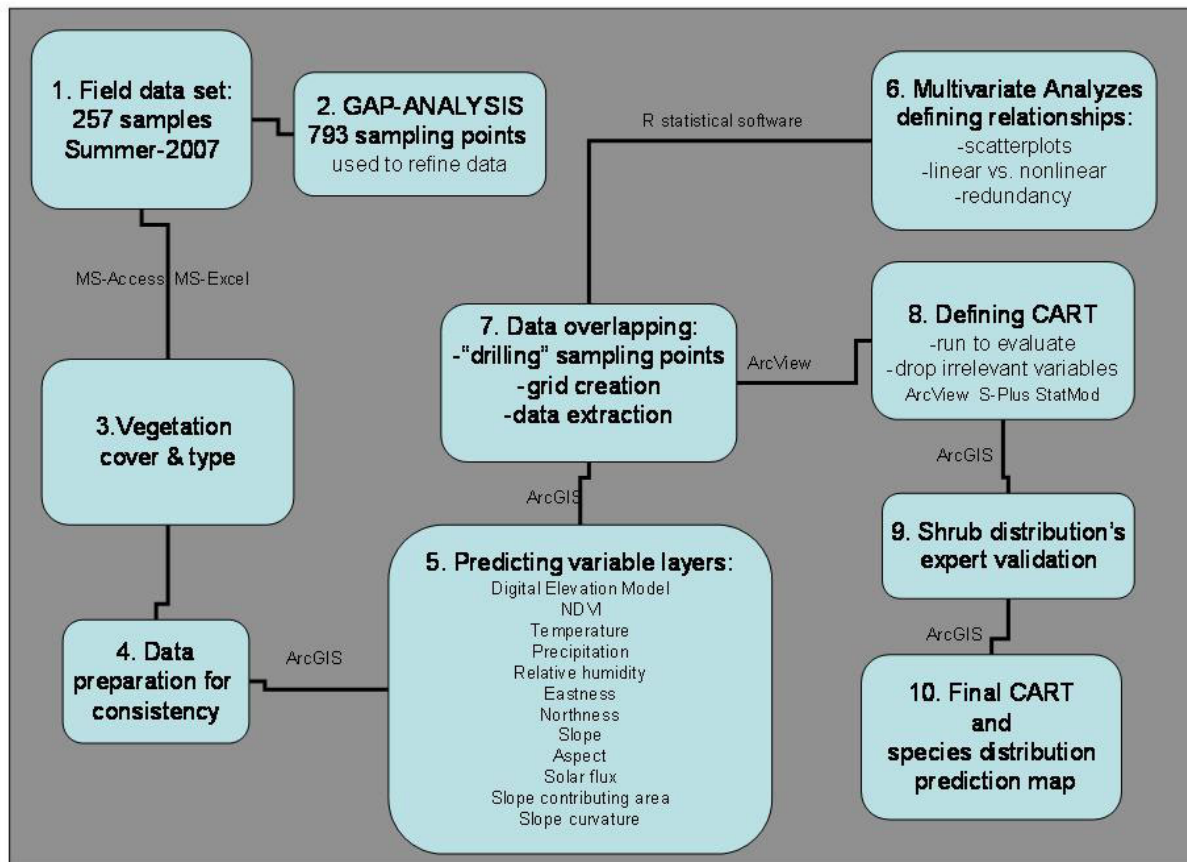


Figure1. Generalized data flow model illustrating creation of CART for shrub species distribution in Rich Co., UT.

Some big sagebrush ecosystems have converted to exotic annual grasslands or to pinion-juniper environments while an equal area has been maintained its natural condition (West 1999).

The area exhibits an ascending elevation gradient (from 1,500 to 2,100 meters above sea level) from East to West. Precipitation may range from 200 to 300 mm per year and temperature will usually range between -40 degrees C to 40 degrees C.

Methodology

The methodology used in this study is described in Figure 1. Field data was acquired in summer of 2007. Field forms were developed in a Microsoft Access database to record GPS coordinates and pictures. Seven shrub species distributed in 257 sites (figure 2) were used as a field-input data in these analyzes (See table 1 for scientific names, common names, and USDA's plant codes). Data was refined and standardized with the SouthWest Gap Analysis data.

Data layers were produced by clipping raw data layers to a 1 km buffered Rich County boundary, and then scaling by standard deviation. The standard deviations were multiplied by 100 and rounded to the nearest whole number. Spatial data was manipulated using ArcGIS ver 9.2, and environmental data was extracted (drilling) from each layer and the Software R was used to study potential relationships, linearity, normality and redundancy among variables. Table 2 shows all explanatory variables used in this study. All layers and data points were arranged in ArcView® ver 3.2 GIS software. Spatial analysis extension was used along with StatMod Zone, an extension for ArcView developed by the USU Remote Sensing and GIS laboratories (Garrard 2003). This extension was designed to simplify statistical modeling with spatial data. This tool facilitates the creation of classification and regression tree (CART) and makes it easy to map the results of these models. The StatMode Zone extension works along with ArcView, and S-Plus to provide the most significant variables and dropping the least relevant variables until it displays the final CART and the species distribution map.

It is important to mention that previous to the use of CART analysis, we used the R statistical software to perform other analyzes such as Principal Component Analysis (PCA) and Cluster Dendrogram Analysis. None of them provided useful results. In addition, the GAP Analysis sampling points for Rich Co. were also used (approximately 900 points). Analyses were also performed using the GAP analysis data alone and combined with the 257 points taken in 2007. The resulting species distribution maps did not provide useful results either. Distribution was confused and did not seem to represent past or current or even a logical species distribution.

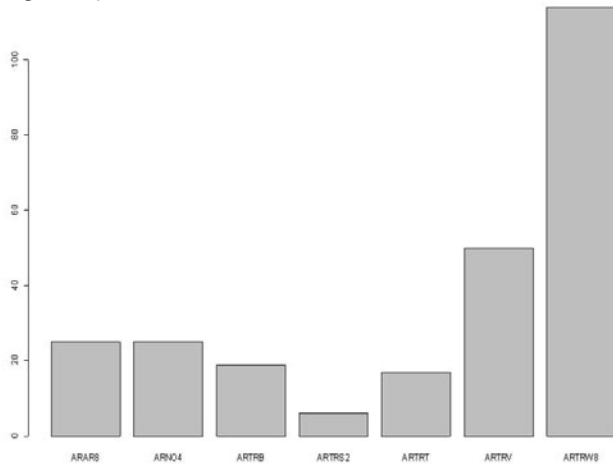


Figure 2. Histogram of number of sampled sites per species. Seven species were sampled in a total of 257 sites - Rich Co., UT.

Model Accuracy

Many methods of accuracy assessment have been discussed in remote sensing literatures (Sardinero 2000, De'Ath 2002). Three measures of accuracy were tested in this study, namely overall accuracy, error matrix and Kappa coefficient. The overall accuracy is evaluated from a predicting model output with respect to geo-referenced data; the term accuracy is used typically to express the degree of 'correctness' of the predicting model (Foody 2002). The matrix error displays the statistics of the image classification accuracy showing the degree of misclassification among classes (Jensen 2005). The Kappa coefficient is a measure of agreement between a model prediction map and reference –field obtained-data (Lowry et al. 2007).

Model accuracy assessment was performed to compute the probability of error for the shrub prediction map. A total of 69 samples (20 percent of

all samples) were previously randomly withheld for the accuracy assessment. Samples were "drilled" into the final prediction map to determine which samples fell correctly into the modeled classes. Procedure involved the use of ArcGIS ver 9.2 and the spatial analysis tool: sampling.

RESULTS AND DISCUSSION

Significant Environmental Variables

Final results showed that the shrub species distribution in the rangelands of Northern Utah, specifically in Rich County, might be driven by precipitation and temperature gradients, and influenced greatly by elevation. Slope-contributing area, NDVI, and solar radiation also resulted in statistical significance, explaining most of the shrub occurrence and distribution. Elevation and eastness were sometimes excluded to avoid redundancy from the analyses, because they presented strong relationships with precipitation and temperature. This analysis provided useful information to study potential relationships, linearity, normality and redundancy among variables, and shows the distribution of shrub species along gradients of all studied environmental variables in Rich County, Utah (figure 3).

In the CART analysis (figure 4), the final model was statistically significant for the following environmental variables: precipitation, temperature, slope contributing area, NDVI and solar radiation. All studied variables and their relationships with the shrub species are described below:

Precipitation: The main driver of presence humidity at each site. For this particular study, Figure 5 shows that the snowfield sagebrush (ARTRS2) sites receive larger amounts of precipitation than sites located at higher elevations. The other species did not seem to receive different amounts of rainfall.

Temperature: shrub species behaved inversely proportional to elevation and precipitation. Figure 6 shows also that snowfield sagebrush (ARTRS2) sites have the lowest average temperature, located at the higher elevation sites. The other species did not seem to be affected by this variable; however, it showed statistical significance at the time of mapping the shrub community distribution.

Table 1. Sagebrush Shrub Species: 7 species or subspecies, 257 sites - Rich Co., UT.

USDA's plant Species code	No. sites	Scientific Name (Genus and Species)	Common Name
ARAR8	25	<i>Artemisia arbuscula</i> subsp. <i>arbuscula</i>	Low sagebrush
ARNO4	25	<i>Artemisia nova</i>	Black sagebrush
ARTRB ^a	19	<i>Artemisia tridentata</i> X " <i>bonnevilleensis</i> "	Boneville sagebrush
ARTRS2	6	<i>Artemisia tridentata</i> subsp <i>spiciformis</i>	Snowfield sagebrush
ARTRT	17	<i>Artemisia tridentata</i> subsp <i>tridentata</i>	Basin big sagebrush
ARTRV ^a	50	<i>Artemisia tridentata</i> subsp <i>vaseyana</i>	Mountain big sagebrush
ARTRW8	114	<i>Artemisia tridentata</i> subsp <i>wyomingensis</i>	Wyoming big sagebrush

^a Plant codes and names are not officially assigned.

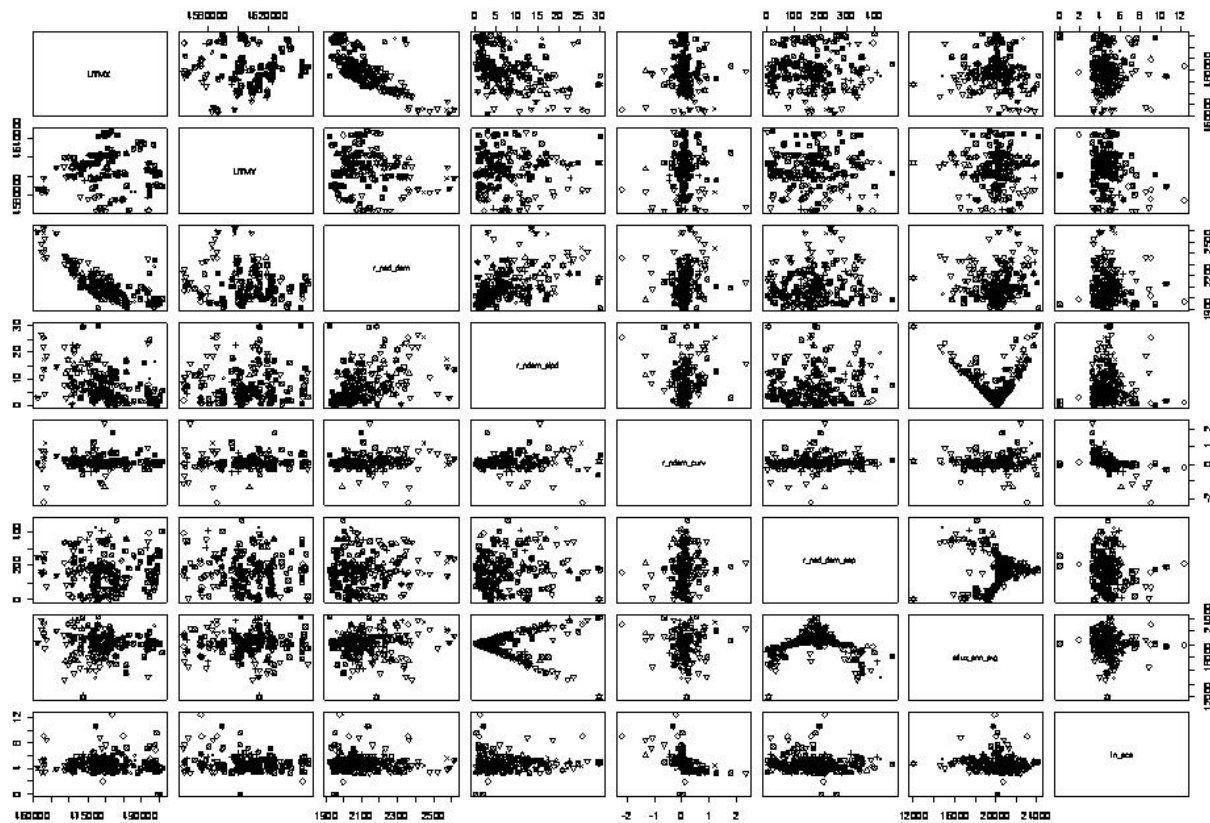


Figure 3. Multivariate assessment of all explanatory variables that explain shrub spatial distribution in Rich County, Utah. Precipitation and temperature are excluded, since they presented strong relationships with elevation and eastness.

Table 2. List of potential explanatory variables used in this study.

Variable	Explanation
Aspect	Aspect, as computed by ArcMap [-1 = flat]
Elevation	Elevation from the USGS National Elevation Data Set (m).
Normalized Difference Vegetation Index (NDVI)	Mean annual NDVI changes over the years for a particular site, a composite of maximum.
Slope curvature	Curvature from r_ned_dem calculated by ArcMap (positive values=convex slope, negative values=concave slope)
Northness	Northing coordinate, NAD83, Zone 12Y UTM coordinates (meters)
Eastness	Easting coordinate, NAD83, Zone 12X UTM coordinates (meters)
Slope	Slope from elevation data set (degrees)
Solar flux index	Annual average solar flux calculated using Zimmerman solar radiation model on r_ned_dem and using Daymet monthly temperature grids (kJ/sq.m/day).
Slope contributing area	log of upslope contributing area calculated using Tarboton "Tau DEM" ArcMap plug-in (ln(m))
Temperature	Average annual temperature calculated from Daymet grids (1/100 C).
Precipitation	Sum of annual precipitation grids calculated from Daymet grids (1/100 cm)

Slope contributing area: this is a measure of moisture availability at each side and it depends on the amount of surface and underground water. Figure 7 shows that there is no apparent change in this variable among the studied shrub species.

NDVI: the Normalized Difference Vegetation Index is an indicator of the amount of greenness reflected by the vegetation. It shows in Figure 8 that there is no apparent difference among species with respect of the greenness values of vegetation.

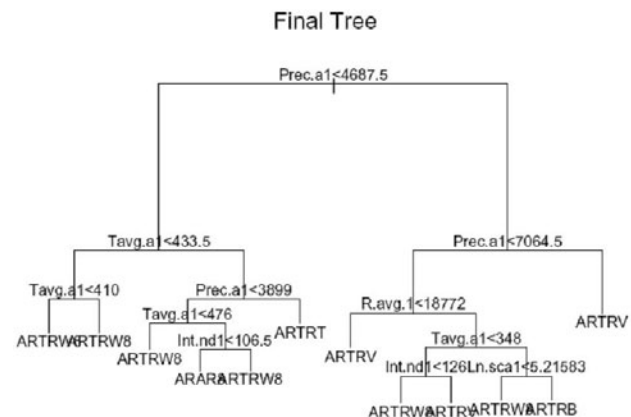
Solar flux index: is a climatic variable that indicates the amount of heat received by a site (figure 9). The species snowfield sagebrush (ARTRS2) was found in areas where solar heat was higher and mountain big sagebrush (ARTRV) was found in areas where solar flux was lower. Solar flux did not appear to be an explanatory variable of the final model.

Elevation: All species were predicted to be found in a range between 1,950 and 2,300 masl (figure 10), except for snowfield sagebrush (ARTRS2) where it can be found at higher altitude between 2,450 and 2,600 masl. The CART analysis did not find this variable to be statistically significant (figure 4).

Slope: All studied shrub species were found to be located within 3 to 17 degrees of slope (figure 11). No major differences were found among species. The CART model did not take into account this variable as a major explanatory variable of the final model (figure 3).

Slope curvature might be a significant topographic variable explaining shrub distribution along rough terrain (figure 12). However, in this study, the CART model dropped this variable due to either not enough number of samples or little consistency in the field information. All species were located on almost flat

surfaces except for black sagebrush (ARNO4), snowfield sagebrush (ARTRS2), and mountain big sagebrush (ARTRV), which were found to occur on slightly concave slopes.

**Figure 4.** Summary of Classification Regression Tree Analysis – S-Plus output.

Aspect is considered one of the most important environmental variables explaining species distribution, because it greatly affects photosynthetic rate and soil moisture availability (figure 13). Most species were found on north facing slopes (60 to 180 degrees) that are cooler, less exposed to the sun heat, and consequently retain more moisture. Aspect was not an explanatory variable of the final CART model (table 3).

Eastness (figure 14) and Northness (figure 15) were also analyzed, and they are associated to the location with respect to the X and Y coordinates of the sampling sites. In the case of the X location, this was associated with elevation, with increases from East to West, and Northness was also associated with temperature, which has a slight decrease moving north. These two variables were not included in the model because of the obvious correlation to the previously mentioned variables.

Table 3. Summary of Classification Tree Analysis of ArcView-SPlus-StatMod output.

Number of branches	Value	Deviance/N	Prediction	Probability
1	7	26.58	ARTRV	0.63
2	8	53.42	ARTRW8	0.76
3	9	107.3	ARTRW8	0.55
4	11	43.49	ARTRT	0.48
5	12	44.12	ARTRV	0.65
6	20	8.04	ARTRW8	0.95
7	42	9.54	ARAR8	0.78
8	43	11.15	ARTRW8	0.71
9	52	23.48	ARTRW8	0.45
10	53	35.87	ARTRV	0.47
11	54	82.46	ARTRW8	0.50
12	55	44.62	ARTRB	0.39

Shrub Community Description

This study is the first to provide an extensive description of shrub vegetation patterns in the Rich County area. We found that shrub vegetation patterns in these shrublands are highly variable and sometimes indistinct, probably more so than in wetter climates. The main finding would probably center on the fact that vegetation composition is ordered along a complex environmental gradient running from the lower to the higher slope gradient. There was also a clear elevation gradient from the valley (east) to the western highest peaks. Within this main gradient, shrub vegetation patterns are further related to specific landforms, topographic positions, microsites, and plant associations.

The environmental features correlated with these shrub distribution patterns are surrogates for the underlying processes and mechanisms. We suggest there are three major drivers of shrub vegetation patterns in Rich County: (i) hillslope processes associated with elevation, (ii) moisture gradients; and (iii) anthropogenic disturbances such as fire and grazing. The distribution of the three locally prevalent subspecies of *A. tridentata* (mountain, Wyoming, and Bonneville sagebrushes) correlates generally with environmental gradients: mountain sagebrush at high elevations, and Wyoming sagebrush and big sagebrush at low elevations. While soil moisture and temperature generally correspond to elevation and aspect, we found that in Rich County, high elevation sites are often too dry for mountain sagebrush (ARTRV), and it is displaced by Wyoming sagebrush (ARTRW8). A hybrid between these two subspecies, Bonneville sagebrush (ARTRB) represents a fourth community type that occurs in habitats that are

intermediate in available moisture. The hybridization zone is clearly delineated at the intermediate elevation, following the contour lines (figure 16). The fifth community type modeled in this study is low sagebrush (ARAR8) a species growing on shallow, fine textured or rocky soils that occur as islands within this region.

Much of the variation in shrub vegetation is a product of hillslope processes and the environmental changes associated with ridge-top to valley bottom gradients. We also suggest that the moisture gradient is one of the main drivers of shrub distribution, and in fact, this is the main driver for most plant community distributions (Parker 1982, Adams and Anderson 1980). It is strongest at the base of the slopes and then decreases as the slope increases. The strength of the gradient may be related to the spatial distribution of precipitation along the elevation axis; that is, there is relatively little precipitation at higher elevations and more precipitation at the valleys. Additionally, shrub distribution is affected by the change in temporal distribution of precipitation, but also to moisture distribution regimes.

Finally, the anthropogenic disturbances have affected the current distribution of shrub vegetation. For instance, species such as black sagebrush (ARNO4) and snowfield sagebrush (ARTRS2) were not mapped because either they do not have enough samples or they did not show a very well defined distribution pattern. For us, the second may be the cause of uneven distribution of these species. Anthropogenic disturbances, such as grazing and fires, are more likely to be the cause of such erratic distribution.

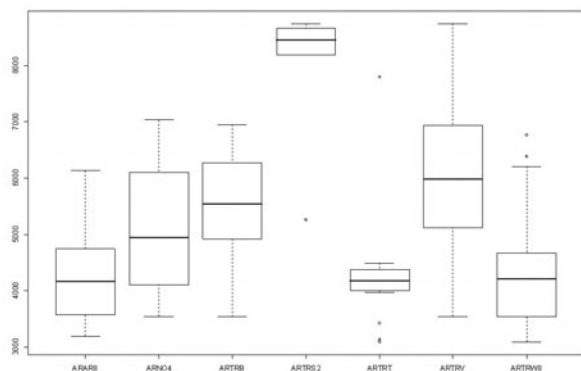


Figure 5. Distribution of shrub species along the Precipitation (1/100cm) gradient, Rich County, Utah.

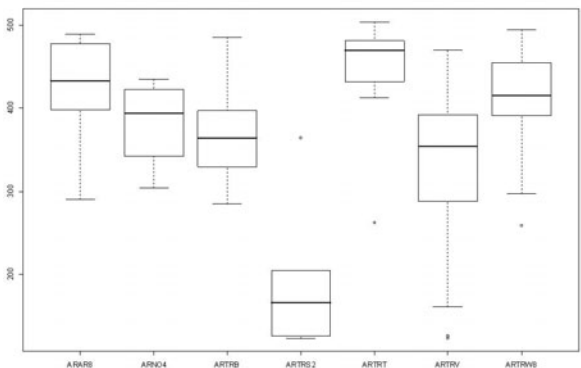


Figure 6. Distribution of shrub species along the Temperature (1/100 degrees C) gradient in Rich County, Utah.

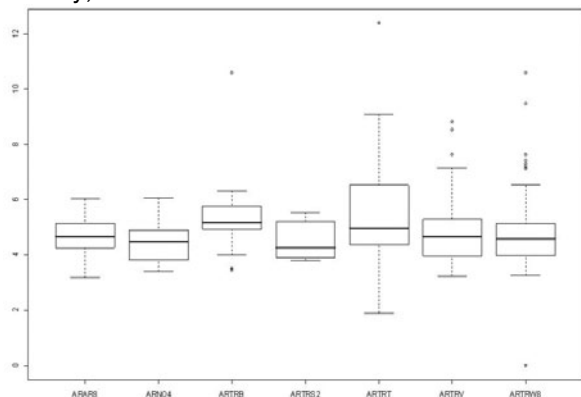


Figure 7. Distribution of shrub species along the Upslope Contributing Area gradient (Log of in meters) in Rich County, Utah.

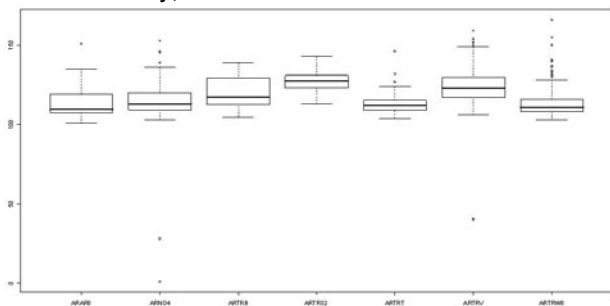


Figure 8. Distribution of shrub species along the NDVI in Rich County, Utah.

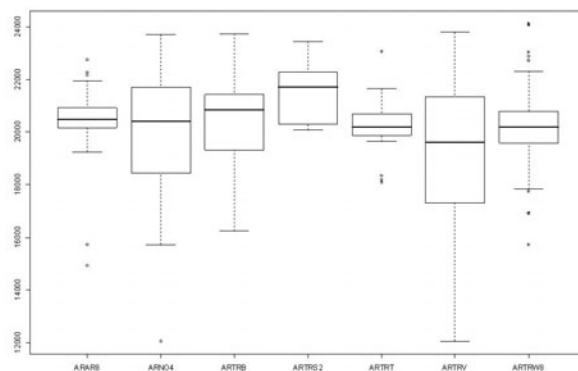


Figure 9. Distribution of shrub species along the solar flux gradient (kJ/sq.m/day) in Rich County, Utah.

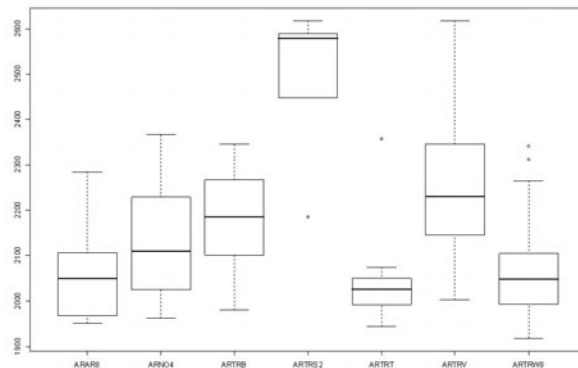


Figure 10. Distribution of shrub species along the elevation (meters) gradient in Rich County, Utah.

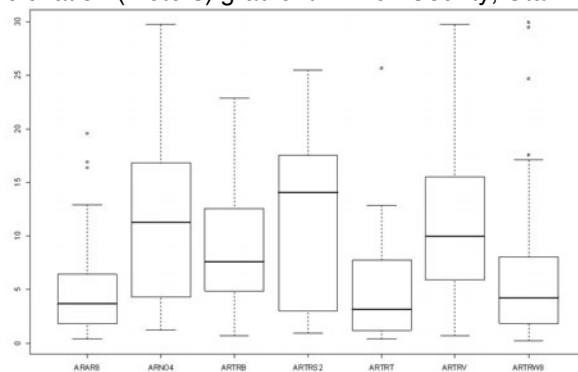


Figure 11. Distribution of shrub species along the slope (degrees) gradient, Rich County, Utah.

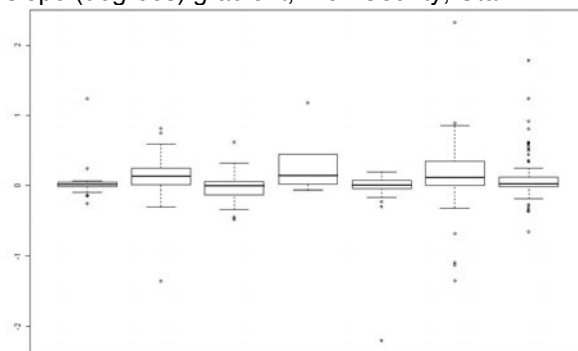


Figure 12. Distribution of shrub species along the slope curvature gradient (Concave (+values) Convex (-values)) in Rich county, Utah.

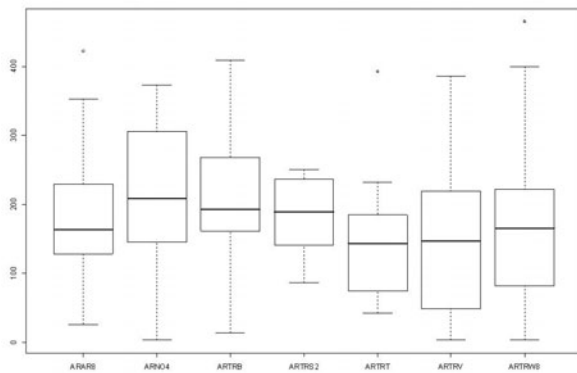


Figure 13. Distribution of shrub species along the aspect gradient (Degrees) in Rich County, Utah.

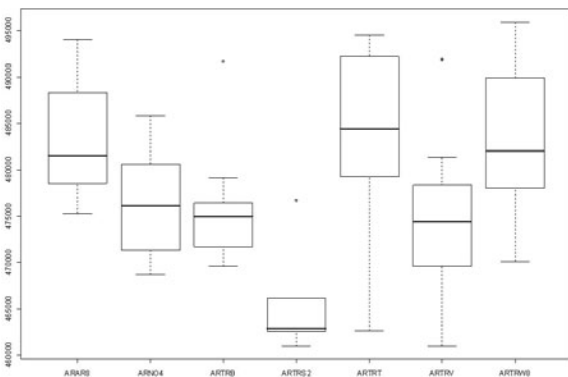


Figure 14. Distribution of shrub species along the X Coordinate (Eastness, in meters) in Rich County, Utah.

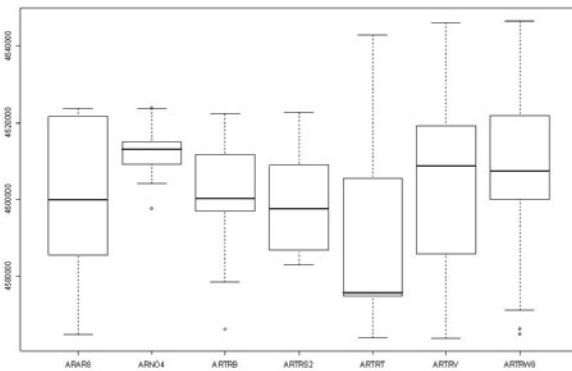


Figure 15. Distribution of shrub species along latitude (Y Coordinates, Northness in meters) in Rich County, Utah.

Shrub Descriptions

Finals results showed that only 5 shrub species (out of seven) were predicted with the final CART model. The spatial distribution of the 5 studied shrub species in a 3-D map of Rich County, Utah can be seen in Figure 16. It shows the distribution of: mountain big sagebrush (ARTRV), Wyoming big sagebrush

(ARTRW8), basin big sagebrush (ARTRT), low sagebrush (ARAR8), and Bonneville sagebrush (ARTRB).

Wyoming big sagebrush (ARTRW8) was the best predicted species and can be found following several branches (rules). Its location can be predicted with the highest probability, 95 percent (branch # 6) (table 3). Black sagebrush (ARNO4) and snowfield sagebrush (ARTRS2) were dropped from the model, because the model either needed more field data or could not establish a distinguishable distribution pattern based on these variables.

Mountain big sagebrush (ARTRV) was predicted at the higher elevation while basin big sagebrush (ARTRT), low sagebrush (ARAR8), and Wyoming big sagebrush (ARTRW8) were predicted at the lower elevations. The proposed hybrid involving Wyoming sagebrush and mountain sagebrush (Shultz 2009) is called "*Bonnevillensis*" (ARTRB), and was predicted in the middle elevation areas, a finding which is consistent with other investigations of hybrid zones for these subspecies of big sagebrush (West 1999, Garrison 2006, Shultz 2009) (Figures 10 and 16). Expert knowledge and the Southwest Regional Gap Analysis Project (Lowry et al. 2005) data were used to corroborate the findings.

This description of species distribution is drawn from a review of literature as it is being compiled in a new work on sagebrush taxonomy and ecology (Tart and Shultz, in prep). These descriptions are supported by our findings of habitat preferences for the various kinds of sagebrush species occurring in Rich County.

1. Mountain big sagebrush (ARTRV). Mountain big sagebrush generally occurs in moister sites than Wyoming sagebrush (ARTRW8), and at higher elevations. In arid mountain ranges, however, the two subspecies may be found at the same elevation. In these situations, mountain sagebrush will be growing in snow-accumulation depressions, east or north-facing slopes, or in areas protected by an overstory of aspen. It occurs in a wide range of mountain habitats, but predominantly on well-drained soils that are higher in organic matter than sites where one typically finds Wyoming big sagebrush.

2. Wyoming big sagebrush (ARTRW8). Wyoming big sagebrush occurs in drier sites than Mountain big sagebrush and is often found on soils with slow infiltration rates (Shumar and Anderson 1986). It also occurs on soils with a greater proportion of summer precipitation (Miller and Eddleman 2000, Winward 2004) or where grazing has reduced the competition from native grasses.

3. Basin big sagebrush (ARTRT). It generally occurs at lower elevations than Mountain or Wyoming big sagebrush and is typically found in valleys. In agricultural areas and low elevation rangelands, this is the subspecies that is now restricted primarily to fencerows and roadsides. It grows in deep, fertile soils that have been plowed for agriculture.

4. Low sagebrush (ARAR8). Low sagebrush occurs on shallow, fine-textured or rocky soils at low to high elevations. It is usually found in isolated "island" communities within the Mountain or Wyoming sagebrush zones.

5. Bonneville sagebrush (ARTRB). Considered a hybrid and named informally as "Bonneville sagebrush" by Al Winward (Garrison 2006, Shultz 2009), this type occurs more commonly with mountain big sagebrush (ARTRV) than with Wyoming big sagebrush (ARTRW8). It has a more diverse herbaceous understory (McArthur and Sanderson 1999, Winward 2004) and is considered an important plant association for various species of wildlife (Shultz 2009).

Model Validation

The Kappa coefficient was $K = 61.26$ percent, and overall accuracy was close to 74 percent (Overall classification = 73.91 percent). The measures of accuracy are shown in Table 4. The overall accuracy is expressed as a percentage of the test-pixels successfully assigned to the correct classes. The results obtained are presented in Table 4, where it contains: the overall confusion matrix, the classification accuracy, and the Kappa coefficient.

From the present analysis, the mountain big sagebrush (ARTRV) achieved 100 percent of classification accuracy with the highest overall accuracy. The 26 sites fell correctly into that class in the predicted model. It was followed by Wyoming big sagebrush (ARTRW8) with 85 percent accuracy, Bonneville sagebrush (ARTRB) (25 percent accuracy) and the low sagebrush (ARAR8) (18 percent accuracy). In general, the model performed better when more field data (reference) was available, but also when the model identified and recognized a clear distribution pattern.

A visual validation was also performed using expert knowledge and field observations. Final distribution of shrub species was corroborated by experts that agreed that final distribution satisfies requirements where the studied shrub species are expected to be found.

Table 4. Error Matrix of the Shrub Prediction Model and the Reference data.

Predicted Data	Reference Data					Total
	ARAR8	ARTRB	ARTRT	ARTRV	ARTRW8	
ARAR8	2	0	0	0	1	3
ARTRB	0	1	0	0	0	1
ARTRT	1	0	0	0	0	1
ARTRV	1	3	0	26	3	33
ARTRW8	7	0	1	0	23	31
Total	11	4	1	26	27	69
% per specie	18.2	25	0	100	85.2	

Overall classification = 73.91%

Kappa Index (K) = 61.26%

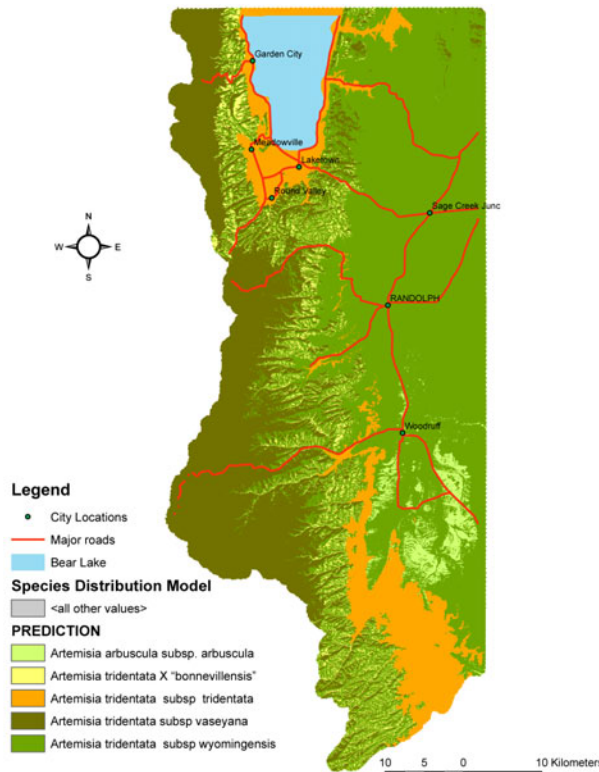


Figure 16. Final map of prediction of shrub species distribution in Rich County, Utah.

CONCLUSIONS

The major findings for this study revealed that environmental features are correlated with patterns associated with mechanisms responsible for shrub distribution in Rich County. The major environmental drivers consisted of processes associated with elevation, temperature, moisture availability and, at small scales by anthropogenic disturbances, such as fire and grazing. This is true, particularly for the most prevalent shrub subspecies of mountain sagebrush (ARTRV), which is usually distributed at higher elevations, and Wyoming sagebrush (ARTRW8) and basin big sagebrush (ARTRT) at low elevations. In Rich County, we also found that higher elevation sites are typically low in moisture availability for mountain sagebrush (ARTRV), and that might be the reason why it is substituted by Wyoming sagebrush (ARTRW8). The Bonneville sagebrush (ARTRB) constitutes a hybrid between these two subspecies and it is the fourth largest shrub community type. Low sagebrush (ARAR8) constitutes the fifth largest shrub community, and its distribution occurs in patches mostly driven by the presence of shallow, fine textured or rocky soils. The actual distribution of black sagebrush (ARNO4) and snowfield sagebrush

(ARTRS2) occurred in our sites, but not in sufficient abundance for predicting modeling or their distributions depend upon, mostly, by human disturbances.

This study demonstrates the effective use of GIS ordination techniques for unbiased identification of homogeneous geographic units, based on topographic, edaphic, and climatic parameters. Older ordination techniques provided little spatial information of where species distribution was located in heterogeneous landscapes. GIS and Remote Sensing techniques along with statistical analyzes, especially CART analysis, offer a promising tool to place plant distributions along environmentally dissected gradients. This analysis would provide important knowledge of where management efforts might be directed to restore this area to its pristine condition.

ACKNOWLEDGEMENTS

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Introducing Big Sagebrush into a Crested Wheatgrass Monoculture

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ABSTRACT

Crested wheatgrass (Agropyron desertorum or A. cristatum) has been effectively used to stabilize arid and semi-arid range sites for decades. Reestablishing native plant materials into these areas is often desirable to increase wildlife habitat and ecological diversity. Due to its competitive nature, efforts to reestablish native plants into crested wheatgrass monocultures have had limited success. Tillage will control the grass but leaves the soil vulnerable to erosion and weed invasion. This publication will report on a trial conducted near Nephi, Utah to find a method of introducing native plants into a crested wheatgrass monoculture without subjecting the resource base to degradation in the conversion process. In this trial, the effect of chemically controlling crested wheatgrass before transplanting big sagebrush (Artemisia tridentata) was studied. Small container grown plants of sagebrush were transplanted either directly into a 60 year-old stand of crested wheatgrass or after chemically controlling the grass. Three different subspecies of big sagebrush; Basin big sagebrush (Artemisia tridentata Nutt. ssp. tridentata), Mountain big sagebrush (Artemisia tridentata Nutt. ssp. vaseyana (Rydb.) Beetle) and Wyoming big sagebrush (Artemisia tridentata Nutt. ssp. wyomingensis Beetle & Young); were planted to see if there would be differences among subspecies. Four years of data indicate that controlling crested wheatgrass prior to transplanting resulted in higher sagebrush survival and faster establishment. There were some differences among sagebrush subspecies. Basin big sagebrush survived equally well with or without grass control but grew faster with grass control. Chemical control of the grass was important for both the survival and growth of Mountain big sage and Wyoming big sage.

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INTRODUCTION

Crested Wheatgrass (*Agropyron cristatum* and *Agropyron desertorum*) has proven its effectiveness as a means to control wind and soil erosion in arid and semi-arid areas over many decades. Its ability to persist is both an asset and a potential hurdle. Once it becomes established the area resources are protected and stabilized from further degradation, but ecological succession may be slowed or halted, depending on the time frame being measured. The ability to establish other plants within crested wheatgrass monocultures is limited at best. Reestablishing native plant materials into these areas is warranted for such purposes as increased wildlife habitat, ecological diversity, and aesthetics. It is possible to consider crested wheatgrass as the beginning of an ecological ladder that stabilizes and protects the resource base. It then can allow transitions to a more diverse community. The methodology used to traverse this ladder has often

resulted in less-than-hoped-for results within expected time frames.

A method to accomplish this transition from a monoculture of crested wheatgrass to a more diverse plant community would be welcomed if the resource base were not subjected to degradation in the conversion process. Tillage of most types (disking, chiseling, plowing, roto-tilling, etc.) to reduce the stand of crested wheatgrass and decrease its competitive effect can result in unacceptable soil erosion. Preservation of the soil stabilizing and weed control benefits of crested wheatgrass is an important issue when considering conversion. Drilling of desired species directly into these stands often meets with failure due to the competitiveness of the grass. Transplanting of small plant materials in containers may provide a method to overcome the initial poor establishment for seed-sown techniques. The potential higher establishment costs associated with transplanting should be measured against the costs

of continued failure or relative low success of seeding techniques. With many sources of restoration funding there is only a one-time opportunity for success. This technique might be useful in the establishment of seed gardens which are often planted as a way to increase the seed bank of desired species in areas of interest.

MATERIALS AND METHODS

Transplants of basin big sagebrush (*Artemisia tridentata* Nutt. ssp. *tridentata*), Mountain big sagebrush (*Artemisia tridentata* Nutt. ssp. *vaseyana* (Rydb.) Beetle), and Wyoming big sagebrush (*Artemisia tridentata* Nutt. ssp. *wyomingensis* Beetle & Young) were obtained from the State of Utah's Forestry, Fire and State Lands; Lone Peak Conservation Nursery; Draper Utah. Plants were grown as containerized seedlings using 3.8 x 21 cm Super Cell Cone-Tainers (Ray Leach Cone-Tainer).

Herbicide treatments (60 year old stand of Nordan Crested Wheatgrass) were completed on April 20, 2004, with 1.75 l/ha of Round-up Ultra (glyphosate). The field was then allowed to lay fallow for a year. Field transplanting was completed on April 7-8, 2005, in both the chemically treated and control treatments. Of the total experimental area (1748 m²) half was treated chemically after dividing into individual treatment blocks (130 m²) each.

Plot location is at the Utah State University Nephi Experiment Station Farm, approximately 6 km south of Nephi, Utah (39° 38' 43" N, 111° 52' 22" W, 1573 m elevation). The Ecological Site designation for the location is: Upland Loam (Mountain Big Sagebrush). Soil at the site is a Nephi Silt Loam (fine-silty, mixed, superactive, mesic Calcic Argixerolls). Mean annual precipitation is 37 cm per year. A randomized complete block design (five replications) was used with twenty-one plants per sub-species planted within each of the treatment blocks. They were arranged in three rows with only the 5 plants in the middle of each species block used for data collection. Inter-transplant spacing was 1.0 m between and within rows.

Survival and plant height was measured in the fall of each year following establishment through 2009. Survival was recorded as a percentage of transplants

still living. Plant height was measured only on live plants. Analysis of variance (Repeated Measures procedure) and mean separations (Least Significant Difference) were accomplished using Statistix 9 (Analytical Software, Tallahassee, Florida). In the analysis of variance, main plot was the subject factor, spray treatment was a between subject factor and sub-species and year were within subject factors.

RESULTS AND DISCUSSION

Means for plant survival and plant height are reported in Table 1. The photos of plots of each sub-species illustrated in Figures 1-6 were taken in June 2009. The spray treatment X sub-species X year interaction was significant for plant survival and plant height and therefore, the three-way interaction means are reported. In other words, the three sub-species of big sagebrush responded differently to the treatments.

Basin big sagebrush was the best adapted to this particular site with 100 percent plant survival in both sprayed and control plots. Plants in the sprayed plots grew rapidly in the first two years after establishment and started leveling off near 100 cm by 2007. In control plots, Basin big sagebrush grew to 28 cm in 2005 and continued to grow through 2009 reaching 58 cm with no indication of a plateau.

Mountain big sagebrush was poorly adapted to the site. Survival was 96 percent in 2005 but dropped off to 68 percent and 12 percent by 2009 in the sprayed plots and control plots, respectively. Where the competition from crested wheatgrass was controlled, plants grew from 18 cm in 2005 to 79 cm in 2008 but declined to 59 cm by 2009. In control plots, plant height was greatest in 2005 at 11 cm. The site was probably too dry for successful establishment of this sub-species even without competition from crested wheatgrass.

Wyoming big sagebrush was intermediate in adaptation with 100 percent survival in the sprayed plots throughout the study. Survival dropped in the control plots from 88 percent in 2005 to 48 percent in 2009. In sprayed plots, plants grew from 23 cm in 2005 to 89 cm in 2009. In control plots, plant height was 19 cm in 2005 and didn't increase throughout the study. Killing the crested wheatgrass prior to planting was critical to the success of Wyoming big sagebrush.



Figure 1. Basin big sagebrush—control.



Figure 4. Basin big sagebrush—sprayed.



Figure 2. Basin big sagebrush—sprayed.



Figure 5. Wyoming big sagebrush—control.



Figure 3. Mountain big sagebrush—control.



Figure 6. Wyoming big sagebrush—sprayed.

Table 1. Plant Survival and Plant Height of Big Sagebrush transplants in a stand of Crested Wheatgrass.

Sub-Species	Year	Plant Survival		Plant Height	
		Control	Sprayed	Control	Sprayed
		%	%	cm	cm
Basin	2005	100	100	28	45
	2006	100	100	29	83
	2007	100	100	37	99
	2008	100	100	48	104
	2009	100	100	58	105
Mountain	2005	96	96	11	18
	2006	64	96	8	45
	2007	64	96	9	73
	2008	52	96	10	79
	2009	12	68	4	59
Wyoming	2005	88	100	19	23
	2006	64	100	11	57
	2007	64	100	12	77
	2008	64	100	18	85
	2009	48	100	17	89
LSD _{0.05}		13		9	

The differential response of the three sub-species of big sagebrush underscores the importance of using adapted plant materials in the conversion of crested wheatgrass lands. In this study, Basin big sagebrush would be the material of choice. The decision to control the wheatgrass with herbicides before planting would depend on the goal of the planting. If the goal was the rapid establishment of patches of sagebrush perhaps to establish seed gardens, then chemical control of the grass could be advantageous. If a more gradual conversion was desired, then transplanting into established uncontrolled stands of grass could be successful with the caveat that success may be more risky in dry years. In either situation, this case study suggests that transplanting containerized plants can be successful.

CONCLUSIONS

Five years of data indicate that controlling crested wheatgrass prior to transplanting resulted in higher sagebrush survival and faster establishment. There were differences between sagebrush subspecies. Basin big sagebrush survived equally well with or without grass control but grew faster with grass control. Chemical control of the grass was important for both the survival and growth of Mountain big sage and Wyoming big sage. The ability to grow viable plant materials in a site long stabilized by a

monoculture of Crested Wheatgrass provide possible evidence of methods to reintroduce native plant materials into our protected rangelands.

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Seasonal Soil CO₂ Flux Under Big Sagebrush (*Artemisia tridentata* Nutt.)

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ABSTRACT

*Soil respiration is a major contributor to atmospheric CO₂, but accurate landscape-scale estimates of soil CO₂ flux for many ecosystems including shrublands have yet to be established. We began a project to measure, with high spatial and temporal resolution, soil CO₂ flux in a stand (11 x 25 m area) of big sagebrush (*Artemisia tridentata* Nutt.) at the Logan, Utah, Forestry Sciences Laboratory. Beginning on Nov. 1, 2009, hourly soil CO₂ flux measurements were made at a single location in the stand using the Li-Cor LI-8100 soil CO₂ flux instrument and 20-cm long-term chamber. Beginning in April, 2010, monthly soil CO₂ flux measurements were made on a grid of 11 locations within the stand using the LI-8100 equipped with the 20-cm survey chamber. Hourly soil temperature (10-cm depth) and volumetric soil water content data were also collected. Soil CO₂ flux, temperature, and water content were highly temporally and spatially variable in the sagebrush stand. Mean (std dev) soil CO₂ flux, temperature, and water content for the measurement period (November 1, 2009 - October 31, 2010) were 0.96 (0.81) $\mu\text{mol m}^{-2} \text{s}^{-1}$, 10.59 (10.11) deg C, and 0.101 (0.062) $\text{m}^3 \text{m}^{-3}$, respectively. Calculated annual soil CO₂ flux obtained by summing all the hourly measurements was 328 g C $\text{m}^{-2} \text{y}^{-1}$. For semi-arid or arid sites where precipitation is less than evapotranspiration, measured total annual soil CO₂ flux will be less than the potential maximum because of dry season suppression of soil respiration when soil water content is very low.*

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INTRODUCTION

Potential changes in the global climate are closely linked with changes in the global carbon (C) cycle (Denman and others 2007). An important part of that cycle is soil respiration returning CO₂ to the atmosphere. Autotrophic (e.g., plant roots) and heterotrophic (e.g., soil microbes) respiration in soils is a major contributor to atmospheric CO₂ and the predominant one in terrestrial ecosystems. Many factors contribute to the high spatial and temporal variability of soil respiration. In the local soil environment, temperature, water content, porosity, atmospheric pressure fluctuations, and substrate quantity and composition control soil CO₂ flux with the atmosphere. At broader landscape scales, soil CO₂ flux varies with overlying plant community types (primary productivity), regional climate, topographic (elevation, aspect) gradients, and geographic (latitude, longitude) gradients that reflect climatic variation. Although soil CO₂ flux data are now available for many biomes (Bahn and others 2010), accurate landscape-scale estimates of soil CO₂ flux

for many ecosystems, including some important shrublands, have yet to be established.

Accurate soil CO₂ flux measurements require costly instrumentation (e.g., Li-Cor LI-8100). To get an accurate annual total soil CO₂ flux at a site, frequent measurements are needed. To obtain accurate landscape-scale estimates of annual total soil CO₂ flux, adequate spatial coverage is needed. Cost constraints and trade-offs between spatial and temporal resolution (Savage and Davidson 2003) drive attempts to find proxies for estimating total annual soil respiration. Examples of such proxy data sets include air temperature and precipitation (Raich and others 2002), litterfall (Davidson and others 2002), primary productivity (Bahn and others 2008), and soil respiration at mean annual temperature (MAT) (Bahn and others 2010).

Big sagebrush (*Artemisia tridentata* Nutt.) is one of the most common and widespread shrub species in the Interior West (McArthur 2000). Given the relatively large land area (50 to 54 million ha as estimated by McArthur and Ott 1996) of sagebrush-dominated

ecosystems, soils under sagebrush may be a large yet relatively unknown contributor to annual soil CO₂ fluxes in the Interior West. To provide some guidance on the spatial and temporal scales needed to more fully characterize soil respiration under big sagebrush, we began a project to measure soil CO₂ flux, temperature, and water content with a high degree of temporal and spatial resolution. Our objectives were to 1) characterize soil respiration under big sagebrush at multiple temporal scales, 2) determine local-scale spatial variability, 3) quantify relationships of soil temperature and water content to soil respiration, and 4) determine N and C substrate limitations on soil microbial respiration.

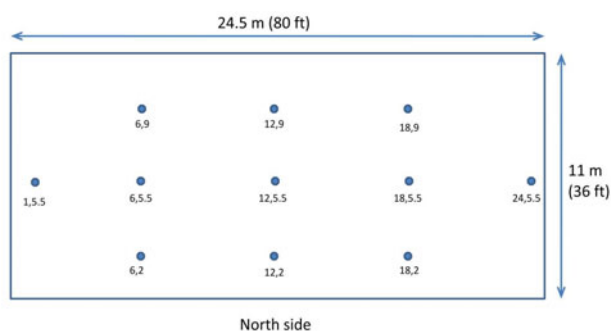


Figure 1. Map of sample points in the sagebrush stand at the Logan Forestry Sciences Laboratory. Numbers refer to distance (m) west and south from lower left corner.

MATERIALS AND METHODS

The sagebrush stand studied (latitude 41° 44.815', longitude 111° 48.453', elevation 1450 m) is on level land at the Logan Forestry Sciences Laboratory (FSL). The Logan FSL is on the ancient Lake Bonneville bench (Provo level) on the west side of the Bear River Range on the eastern edge of the Great Basin. Mean annual air temperature is 8.9 deg C and mean annual precipitation is 450 mm (1893-2005) (data from Utah State University Climate Center). The stand measures 11 m x 25 m (0.0275 ha) and was planted in the mid-1990s in a site formerly occupied by desert wheatgrass (*Agropyron desertorum* (Fisch. ex Link) Schult.). Today the site consists entirely of a closed (no significant gaps) big sagebrush canopy (0.5 to 2 m height) with some minor forb and grass (chiefly cheatgrass (*Bromus tectorum* L.)) components along the edges.

A grid of 11 sample points was established within the stand for soil core sampling and soil respiration

measurements under the sagebrush canopy (figure 1). A PVC soil respiration collar (20-cm diameter x 11.5-cm in height) was installed adjacent to each sample point leaving about half the collar height above ground. The above-ground median height of each soil collar (known as the chamber offset) was determined by measuring the soil collar height at the north, south, east, and west edges and in the middle (from soil surface to a straight edge placed across the top of the soil collar) and taking the median of the 5 readings.

Temporal Soil CO₂ Flux

To measure temporal variability of soil CO₂ flux, the Li-Cor LI-8100-104 long-term chamber (Li-Cor, Lincoln, Nebraska) was installed at the west end of the sagebrush stand with the collar under the sagebrush canopy. The long-term chamber was connected to the LI-8100 automated control unit (ACU), which houses the infra-red gas analyzer (IRGA) that measures changing CO₂ efflux from the soil. The ACU was housed in an instrument shelter. The system was powered by three 98 A/hr Pb-acid gel-cell batteries connected in series. These were re-charged with two 40-watt solar panels.

Beginning at midnight on November 1, 2009, hourly soil CO₂ flux readings were collected for an entire year and continue to be collected. There were some brief periods (from hours up to about 2 days) during winter months when solar re-charge was insufficient to power the system. In such cases, we waited until battery re-charge was sufficient to operate the system. Occasional data gaps during summer months (up to 2 days) were due to operation of the LI-8100 with the survey chamber at other forest, shrub, and meadow ecosystems. In all, 7211 hourly readings were collected from November 1, 2009 through October 31, 2010. The IRGA was calibrated monthly using zero air (zero CO₂ and H₂O content) for baseline calibration and 1000 umol CO₂/mol air for the span calibration. Drift was typically less than 5 umol/mol CO₂ per month.

Annual total soil CO₂ flux was obtained by summing all the measured and interpolated (for missing values) hourly values from November 1, 2009 through

October 31, 2010. To obtain this value, total values for each hour were calculated by multiplying the LI-8100-calculated soil CO₂ flux in $\mu\text{mol CO}_2 \text{ flux m}^{-2} \text{ s}^{-2}$ by 3600 seconds in an hour. Then after summing all the measured and interpolated hourly values, total μmol of CO₂ for the year was converted into the more commonly reported grams of C.

Ancillary soil temperature and soil moisture data also were collected on an hourly basis during long-term chamber operation. Soil temperature at 5-cm depth was collected with the soil temperature probe included with the LI-8100. A separate channel logged soil temperature at a 10-cm depth using a thermistor connected to the LI-8100 interface box. Volumetric soil moisture content (8-cm depth) was measured with a Delta-T ML2x thetaprobe (frequency domain type capacitance sensor) connected to the LI-8100 interface box.

Spatial Soil CO₂ Flux

Spatial soil CO₂ flux was measured monthly at all 11 soil collars beginning in April, 2009 using the LI-8100 20-cm survey chamber and ACU. Soil temperature (5-cm depth) and soil moisture (8-cm depth) were also measured adjacent to the spatially distributed collars.

Soil Sampling and Characterization

Soil cores (5-cm diameter x 10-cm depth) were collected at each sample point in November, 2009 to provide soil characterization information. Soil probe (1-cm diameter x 10-cm depth) samples were collected monthly on days when survey-chamber soil CO₂ fluxes were measured.

Soil cores were air-dried at ambient temperature. A subsample was analyzed for gravimetric residual moisture content by oven-drying at 105 C and soil-core bulk density was calculated. The air-dried soil samples were then sieved through a 2-mm stainless-steel screen and the less-than and greater-than 2-mm fractions were weighed. The <2-mm fraction was analyzed for particle-size distribution (sand, silt, and clay) (Klute 1986), soil pH, total organic and inorganic C and total N by Leco TruSpec and RC-412 combustion analyzers, and 2 M KCl-extractable NH₄-N and NO₃-N by flow-injection colorimetric analysis (Sparks 1996).

Microbial Respiration by BD Oxygen Biosensor (BDOBS)

Oxygen consumption by native soil microbes was measured with a 96-well microplate platform containing an O₂-sensitive fluorophore (4,7-diphenyl-1,10-phenanthroline ruthenium chloride) in a silicone gel matrix permeable to O₂ (BD Biosciences, Franklin Lakes, New Jersey). The fluorescence of the ruthenium dye is quenched by O₂. The fluorescence signal from the fluorophore-gel complex embedded in the bottom of microplate wells increases in response to respiration (O₂ consumption) in overlying microbial samples (e.g., soil slurries). Results are expressed in normalized relative fluorescence units (NRFU) calculated as the ratio of fluorescence of soil sample plus any amendments in each well to the fluorescence of each empty well. Microbial respiration in response to nutrient limitations, substrate composition, or other environmental influences can be measured with this method. In this study, microbial respiration in soil slurries (1: 2.5 ratio of soil to sterile deionized water) was measured. Microbial response to added N (10 mg/L NH₄-N from (NH₄)₂SO₄), C substrate (50 mg/L C from glucose), and added N + C was also measured (Zabaloy and others 2008). Control wells contained amendment solutions (sterile deionized water, (NH₄)₂SO₄, glucose) but no soil.

RESULTS

Soil Characteristics

The top 10 cm of soil under the sagebrush stand is sandy with about 20 percent coarse fragment content (table 1). The bulk density is considered optimal for growing plants. This soil is highly calcareous with a strongly alkaline pH and total inorganic C (from carbonates) exceeds total organic C. At the end of the growing season in November, the soil had concentrations of available N below 10 mg kg⁻¹, mostly as NO₃-N.

Hourly Soil CO₂ Flux, Temperature, and Water Content

Early November, 2009, was relatively warm and dry and hourly soil CO₂ flux generally exceeded 0.5 $\mu\text{mol m}^{-2} \text{ s}^{-1}$ (figure 2). In late November the top 10 cm of soil began to freeze (soil temp decreased to about 0 deg C), soil water content decreased as free water in the soil profile froze, and soil CO₂ flux decreased overall. The top 10 cm of soil remained frozen throughout December, January, and February, and soil respiration slowly declined and eventually ceased

when no measureable soil respiration was recorded throughout February. In early March, soil water content sharply increased as the soil surface began to thaw, although overall surface soil temperature remained at freezing. A corresponding large increase in soil CO₂ flux was associated with soil surface thawing. Throughout spring, soil water content remained high via replenishment from spring storms. As the soil slowly warmed, hourly soil CO₂ flux generally increased, but hourly flux showed large changes on most days. With arrival of the dry season in July, soil temperature continued to warm, but soil water content decreased substantially. Soil CO₂ flux decreased along with the decreasing soil water content. Major storms in mid-June, early September, and mid and late October produced large increases in soil water content and attendant pulses of soil respiration. Annual total soil CO₂ flux obtained by summing all the measured and interpolated (for missing values) hourly values was 328 g C m⁻² y⁻¹.

Table 1. Physical and chemical properties of the sagebrush stand soil (0-10 cm). Values shown are the means $\pm$ std errs of 11 soil cores collected at the sample points shown in figure 1.

Soil property	mean $\pm$ std err
Bulk density, g/cm ³	1.28 $\pm$ 0.05
Coarse fragments, %	20.0 $\pm$ 1.8
Sand, %	58.7 $\pm$ 2.3
Silt, %	30.3 $\pm$ 1.8
Clay, %	11.0 $\pm$ 0.7
Soil pH	7.80 $\pm$ 0.03
Total organic C, %	1.97 $\pm$ 0.13
Total inorganic C, %	2.95 $\pm$ 0.17
Total C, %	4.92 $\pm$ 0.16
Total N, %	0.151 $\pm$ 0.012
2 M KCl-extractable NH ₄ -N, mg/kg	2.2 $\pm$ 0.5
2 M KCl-extractable NO ₃ -N, mg/kg	6.3 $\pm$ 2.9

Daily Mean Soil CO₂ Flux, Temperature, and Water Content

Hourly soil CO₂ flux can be highly variable throughout any given day (figure 2) and tend to be more erratic than soil temperature which is associated with daytime heating and nighttime cooling. To dampen the amplitude of the measured quantities versus time plots, daily mean $\pm$ std err values of soil CO₂ flux, temperature, and water content were calculated (figure 3). Day to day changes throughout the seasons are more easily discerned with daily means plots. Daily mean soil CO₂ flux ceased (February) after the soil was frozen for an extended period (December through February). Soil CO₂ flux increased linearly with soil temperature during the spring and early summer up to about 20 deg C, but then decreased with continued soil warming as soil

drying occurred (figure 4). Thus a peak shaped distribution relationship between soil CO₂ flux and soil temperature was found.

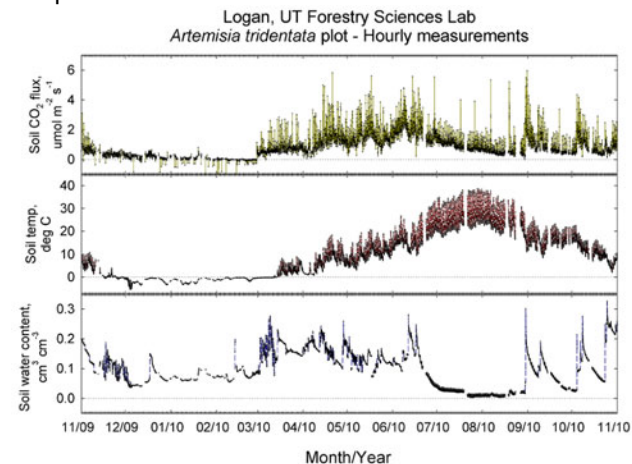


Figure 2. Hourly soil CO₂ flux (top), temperature (middle), and water content (bottom) from November 1, 2009 through October 31, 2010 in a sagebrush stand at the Logan Forestry Sciences Laboratory.

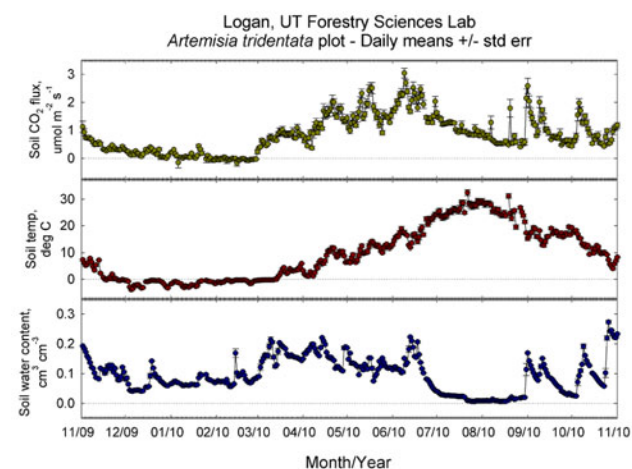


Figure 3. Daily mean $\pm$ std err soil CO₂ flux (top), temperature (middle), and water content (bottom) from November 1, 2009, through October 31, 2010, in a sagebrush stand at the Logan Forestry Sciences Laboratory.

The relationship between soil CO₂ flux and soil water content is more complex. In general, soil respiration was greater with increasing soil moisture (figure 4), but there was a hysteresis effect. Pulses of CO₂ flux were seen during and immediately after passing rainstorms (e.g., mid-June, early September, mid and late October) as soil microbial activity was stimulated (figure 3). As the soil dried, soil respiration again decreased. At times, soil water content during wetting had a different associated soil CO₂ flux value than soil CO₂ flux at the same numerical water content value during drying.

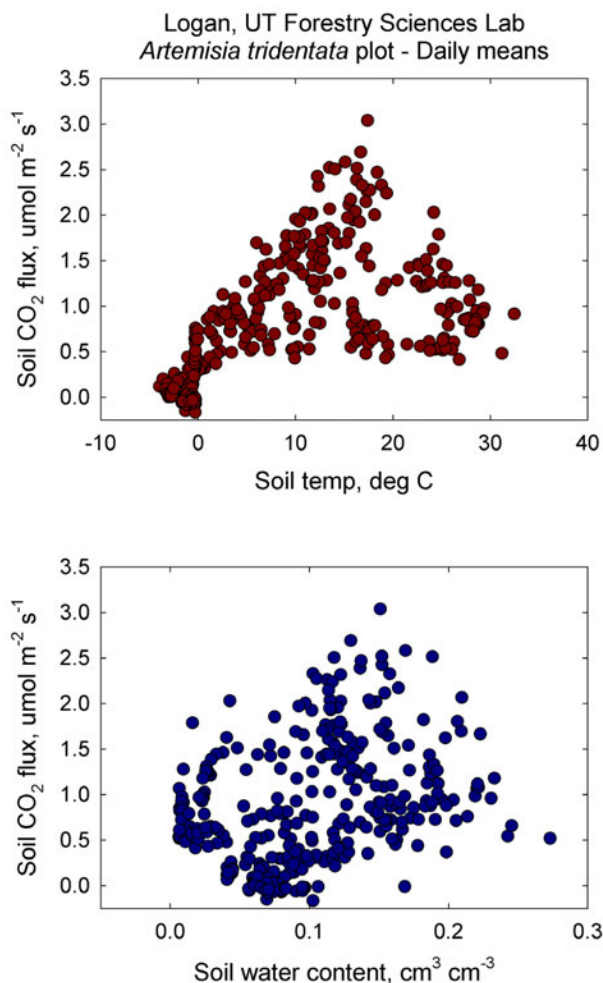


Figure 4. Relationship between daily mean soil CO₂ flux and daily mean soil temperature (top) and between daily mean soil CO₂ flux and daily mean soil water content (bottom) from November 1, 2009, through October 31, 2010, in a sagebrush stand at the Logan Forestry Sciences Laboratory.

Monthly Mean Soil CO₂ Flux, Temperature, and Water Content

Seasonal changes in mean soil CO₂ flux, temperature, and water content are apparent in the monthly mean plots (figure 5). Even though July and August mean soil temperatures were higher than that in June, soil CO₂ flux was less because the soil was drier. The soil remained frozen throughout December, January, and February in the 2009-2010 winter season with no significant thaw period. As a result, mean soil CO₂ flux entirely ceased during February. Thus, a two-month lag was observed between onset of soil freezing and cessation of measurable soil respiration.

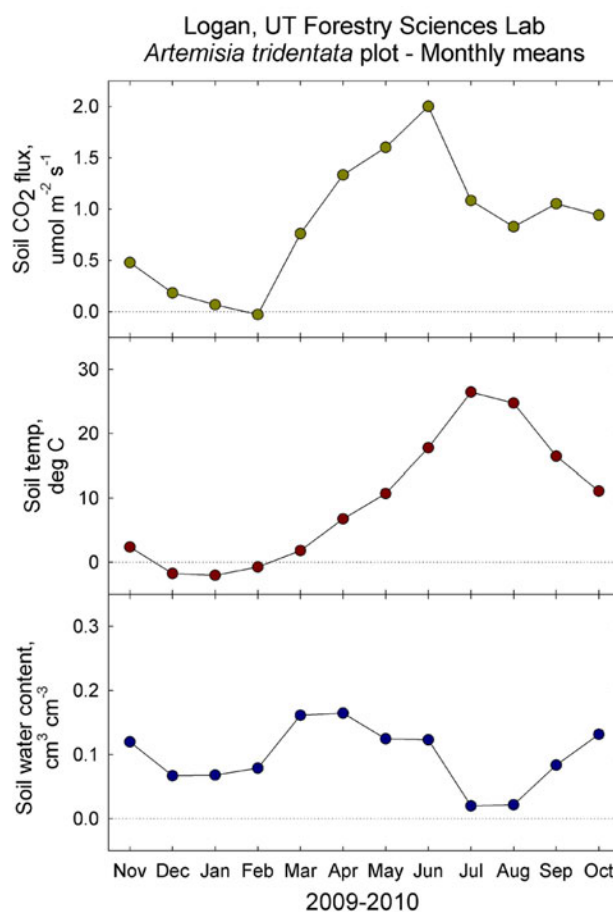


Figure 5. Monthly mean $\pm$ std err soil CO₂ flux (top), temperature (middle), and water content (bottom) from November, 2009, through October, 2010, in a sagebrush stand at the Logan Forestry Sciences Laboratory.

Seasonal Mean Soil CO₂ Flux, Temperature, and Water Content

Mean nighttime soil CO₂ fluxes were slightly less (0.05 to 0.16 $\mu\text{mol CO}_2 \text{m}^{-2} \text{s}^{-1}$) than those during daylight hours during fall and winter (table 2). During spring and summer, mean nighttime soil CO₂ flux averaged about 0.35 $\mu\text{mol CO}_2 \text{m}^{-2} \text{s}^{-1}$ less than the daytime mean. Nighttime soil CO₂ flux averaged 78, 82, 78, and 72 percent of daytime values for the fall, winter, spring, and summer seasons, respectively. The decrease in nighttime CO₂ flux compared to daytime was probably related to soil temperature differences since there was no difference in mean soil water content between daylight and nighttime hours. Daytime and nighttime soil temperature contrasts in terms of absolute temperature differences were greater in spring, summer, and fall than in winter (1.2 to 1.5 $^{\circ}\text{C}$ in spring, summer, and fall versus only 0.3 $^{\circ}\text{C}$ in winter). On a relative basis, nighttime soil

temperatures averaged 82, 63, 85, and 94 percent of daytime values for the fall, winter, spring, and summer seasons, respectively.

Overall, mean soil CO₂ flux under sagebrush was found to be highest during the spring when the soil water content was relatively high and the soil temperature was high enough to promote root respiration and vigorous soil microbial activity in terms of organic substrate decomposition (table 3). Mean soil temperature more than doubled in summer compared to spring, but mean soil water content was less than a third as much leading to a significant decrease in mean soil respiration.

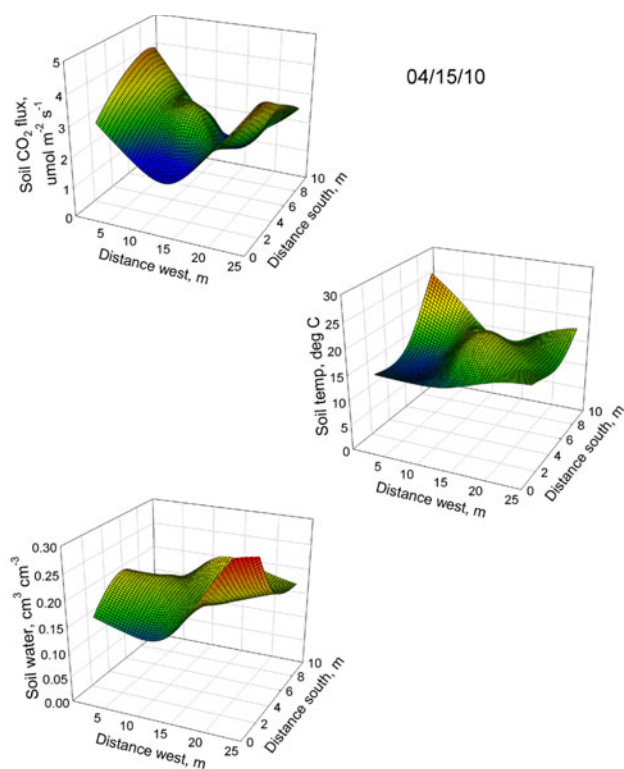


Figure 6. Spatial distribution of soil CO₂ flux (top), temperature (middle), and water content (bottom) on April 15, 2010, in a sagebrush stand at the Logan Forestry Sciences Laboratory.

Spatial Distribution of Soil CO₂ Flux, Temperature, and Water Content

Soil CO₂ flux, temperature, and water content were spatially variable within the 11 x 25 m sagebrush stand and the extent and complexity of spatial variability changed with the seasons (figures 6 and 7). In the spring (e.g., on April 15, 2010), soil respiration was highest at the east and west ends of the stand, soil temperature was highest in the

southeast corner, and soil water content was highest in the northwest corner. In the summer (e.g., on August 16, 2010), soil respiration tended to be uniformly low, soil temperatures were high throughout the plot, and soil water content was low throughout the plot. Overall, the west side of the stand was wettest during spring months, but during the summer, the east side tended to remain the wettest as the west side dried.

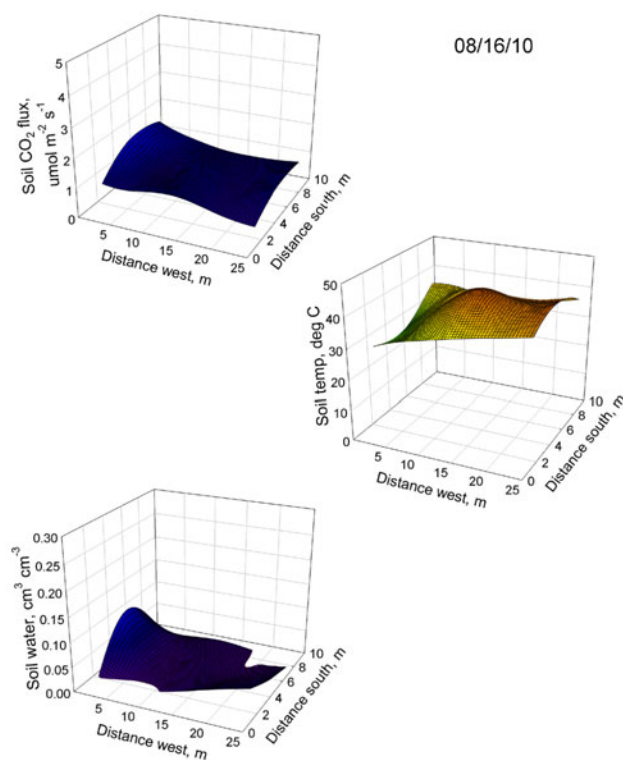


Figure 7. Spatial distribution of soil CO₂ flux (top), temperature (middle), and water content (bottom) on August 16, 2010, in a sagebrush stand at the Logan Forestry Sciences Laboratory.

Boxplots of monthly soil CO₂ flux, temperature, and water content for the 11 sample points in the stand reveal how the magnitude of spatial variability changed through the seasons (figure 8). The spatial distribution of soil temperature tended to follow a statistically normal distribution each month. The contrast between highest and lowest temperatures was greatest during July and August. Soil water content spatial distribution was more skewed during the summer since some points in the stand were wetter than most of the other points. Although soil CO₂ flux tended to be spatially normally distributed in the spring, outlier values skewed the statistical distribution in the summer.

Table 2. Seasonal daytime and nighttime mean $\pm$ std err soil CO₂ flux, temperature, and water content. Because hourly measurements did not begin until November 1, 2009, the fall season data includes November 1 through December 20, 2009 and September 23 through October 31, 2010.

2009-10 Season	Photoperiod	Soil CO ₂ flux, umol m ⁻² s ⁻¹	Soil temperature, deg C	Soil water content, m ³ m ⁻³
			mean $\pm$ std err	
Fall	Day	0.71 $\pm$ 0.02	6.77 $\pm$ 0.26	0.106 $\pm$ 0.002
	Night	0.55 $\pm$ 0.01	5.56 $\pm$ 0.21	0.108 $\pm$ 0.002
Winter	Day	0.28 $\pm$ 0.02	-0.48 $\pm$ 0.07	0.108 $\pm$ 0.002
	Night	0.23 $\pm$ 0.01	-0.76 $\pm$ 0.05	0.103 $\pm$ 0.002
Spring	Day	1.70 $\pm$ 0.03	10.33 $\pm$ 0.16	0.146 $\pm$ 0.001
	Night	1.33 $\pm$ 0.02	8.81 $\pm$ 0.16	0.146 $\pm$ 0.001
Summer	Day	1.21 $\pm$ 0.02	23.73 $\pm$ 0.20	0.043 $\pm$ 0.001
	Night	0.87 $\pm$ 0.02	22.20 $\pm$ 0.18	0.042 $\pm$ 0.002

Soil Microbial Respiration

Native soil microbial respiration in the sagebrush stand expressed as peak oxygen consumption (normalized relative fluorescence units or NRFU) was high (figure 9). Without adding any additional nutrients and relying strictly on native substrate C levels, native soil microbe communities showed an almost six-fold increase in peak oxygen consumption over control wells (no soil), which had an NRFU value of 1. Adding 10 mg/L extra N (from (NH₄)₂SO₄) produced a slight increase in microbial respiration indicating that this sagebrush stand soil is not N deficient for microbial utilization. Adding extra C substrate (50 mg/L C as glucose), produced a large increase in microbial respiration compared to unamended soil (more than 8-fold the O₂ consumption compared to controls). Adding N and C together did not increase O₂ consumption more than the added glucose alone. Although there is adequate C substrate in the soil for the native microbial communities, an additional positive response was obtained to added substrate (more food in the form of glucose).

DISCUSSION

Soil CO₂ flux data for sagebrush-dominated areas of the Great Basin and adjacent physiographic areas are sparse and limited to select sites and years. Furthermore, most estimates of soil CO₂ flux in this region are based on net ecosystem exchange (NEE) measurements rather than chamber-based methods. For example, Gilmanov and others (2004) measured NEE of CO₂ using the Bowen ratio energy balance method during winter (November 1 - March 15) at two sagebrush sites in Idaho and one in Oregon. During winter months, autotrophic respiration is at or very near zero so NEE would tend to be heterotrophic CO₂ respiration from soil. They obtained mean (std dev)

daily CO₂ flux values of 0.68 (0.56) (Burns, Oregon, in 2000/2001), 1.23 (1.19) (INEEL, Idaho, in 1999-2001), and 1.31 (0.80) (Dubois, Idaho, in 2000) g CO₂ m⁻² day⁻¹. Our mean chamber-based measurement of soil CO₂ flux for the same November to March time period in 2009-2010 was 0.32 umol CO₂ m⁻² s⁻¹, which is equivalent to a daily mean of 1.22 g CO₂ m⁻² day⁻¹, a value very much in line with the wintertime Bowen ratio estimated values for sagebrush-steppe areas given by Gilmanov and others (2004). Gilmanov and others (2004) also summarized average wintertime soil CO₂ fluxes for a range of biomes from published values. These values ranged from a low of 0.23 g CO₂ m⁻² day⁻¹ in an arctic tundra in ALASKA to a high of 4.4 g CO₂ m⁻² y⁻¹ for a lowland grassland in Switzerland. Wintertime soil CO₂ flux values in sagebrush-steppe soils from the Gilmanov and others (2004) NEE estimates and our chamber-derived value are within the range of wintertime values for other biomes.

In a different plant community type (cheatgrass) in southwest Idaho, on the boundary between the Snake River Plain and Great Basin, Myklebust and others (2008) measured a combined NEE estimate from eddy covariance, soil CO₂ gradient, and soil chamber (LI-8100) methods for annual soil CO₂ flux during 2005 of 406 $\pm$ 73 g C m⁻². Our annual soil CO₂ flux under sagebrush during 2009-2010 was 328 g C m⁻² y⁻¹, which is similar, but a little lower than the cheatgrass site in Idaho. Differences in vegetation type, soil properties, and climatic conditions during the measurement periods could account for the relatively small difference between these annual totals.

Bahn and others (2010) summarized calculated annual total soil CO₂ flux for 15 Mediterranean, subhumid, and semi-arid forests, savannas,

shrublands, and grasslands. These ranged from 345 g C m⁻² y⁻¹ for intershrub microsites in a semi-arid Mediterranean shrubland with prickly burnet (*Sarcopoterium spinosum* (L.) Spach) to 1456 g C m⁻² y⁻¹ for a holly oak (*Quercus ilex* L.) forest. The mean (std dev) annual total soil respiration for all 15 biomes was 684 (68) g C m⁻² y⁻¹. Mean annual soil temperature (MAT) in these 15 biomes ranged from 8.5 to 22.9 deg C while mean annual precipitation (MAP) ranged from 280 to 844 mm. Most of these biomes are wetter and warmer than our sagebrush site so it is not surprising that they had higher annual total soil CO₂ flux values. The biome with annual total soil respiration closest to our calculated value (328 g C m⁻² y⁻¹) was a semi-arid Mediterranean prickly burnet shrubland (345 g C m⁻² y⁻¹). Relative to our sagebrush site, this prickly burnet shrubland was warmer (MAT = 22.9 deg C), which favors increased soil respiration, and drier (MAP = 300 mm), which favors respiration inhibition. In contrast, the biome with the closest MAT (10.4 deg C) and MAP (460 mm) to our site had an annual total soil respiration more than double that of our site (726 g C m⁻² y⁻¹). This was a Canary Island pine (*Pinus canariensis* C.Sm.) forest on the island of Tenerife, Spain and most likely has a much thicker forest floor layer to provide more substrate for heterotrophic respiration during organic matter decomposition.

Table 3. Seasonal mean $\pm$ std err soil CO₂ flux, temperature, and water content. Because hourly measurements did not begin until November 1, 2009, the fall season data includes November 1 through December 20, 2009 and September 23 through October 31, 2010. Total annual soil CO₂ flux was 328 g C m⁻² s⁻¹.

2009-10 Season	Soil CO ₂ flux, umol m ⁻² s ⁻¹	Soil temperature, deg C	Soil water content, m ³ m ⁻³
mean $\pm$ std err			
Fall	0.62 $\pm$ 0.01	6.07 $\pm$ 0.16	0.107 $\pm$ 0.002
Winter	0.25 $\pm$ 0.01	-0.64 $\pm$ 0.04	0.105 $\pm$ 0.001
Spring	1.55 $\pm$ 0.02	9.69 $\pm$ 0.012	0.146 $\pm$ 0.001
Summer	1.07 $\pm$ 0.02	23.09 $\pm$ 0.14	0.043 $\pm$ 0.001
2009-10	0.96 $\pm$ 0.01	10.59 $\pm$ 0.12	0.101 $\pm$ 0.001

Autotrophic respiration by sagebrush roots would occur only during the growing season (about 7 months) for this species at our location and is temperature related (Bahn and others 2010). Heterotrophic respiration is strongly controlled by soil

temperature, moisture, and substrate availability in semi-arid soils (Conant and others 2004). Wintertime soil respiration is related to freeze/thaw cycles and wind events (Gilmanov and others 2004). During the 2009-2010 winter season at our site, the typical wintertime freeze/thaw cycles that often occur here were not observed since the shallow soil froze in early December and remained frozen until early March (figure 2).

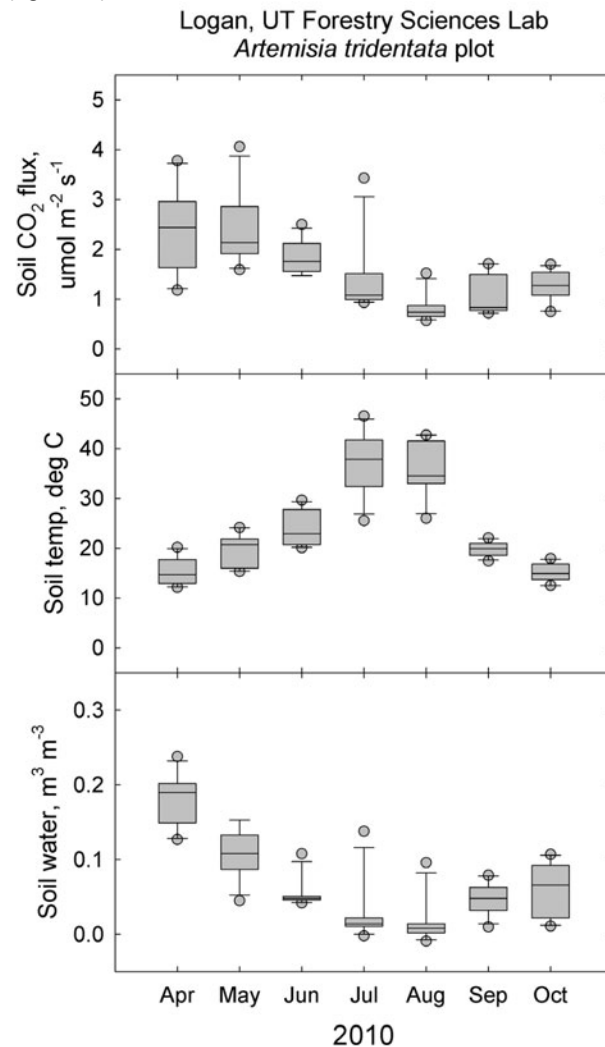


Figure 8. Boxplots of monthly (April through October, 2010) soil CO₂ flux (top), temperature (middle), and water content (bottom) for all 11 sample points in a sagebrush stand at the Logan Forestry Sciences Laboratory. Median spatial values are shown as a horizontal line surrounded by a box representing the 25 and 75 percentiles. Error bars represent the 10 and 90 percentiles and outliers are shown as points.

Aside from temperature effects, there are also resource limitations on heterotrophic respiration. The main resource limitations for heterotrophic soil

respiration under sagebrush at our location were water and C substrate. During or just after every spring through fall precipitation event, a significant, sometimes large increase in soil respiration was observed (for example, figure 3). The more immediate increase was probably due to a stimulation of microbial respiration in the upper few centimeters of soil. As water infiltrated the soil, respiration from plant roots deeper in the soil would then be stimulated. As the soil dried following a precipitation event, microbial respiration declined and perhaps even ceased at very low water contents. The stimulatory effect of summertime rainfall on soil respiration in semi-arid and arid plant communities is well known (for example, Austin and others 2004; Jin and others 2007, 2009; Sponseller 2007; Xu and others 2004).

Although many studies have attempted to separate root and microbial respiration in situ (e.g. Hanson and others 2000), we did not do so in this study since distinguishing between the two sources is blurred by the inseparable root-microbe associations within the rhizosphere (Baggs 2006). Nevertheless, the rapid response of soil respiration to the spring thaw and summer and fall precipitation events indicates a strong heterotrophic contribution to overall soil respiration. To provide additional insight on factors controlling heterotrophic respiration in sagebrush soils, we used the BD oxygen biosensor method to study C and N resource limitations on microbial respiration. Adding C substrate greatly stimulated microbial respiration, but adding extra N did not. This finding indicated that energy, not N, was more limiting to heterotrophic respiration in this soil (figure 9). The lack of microbial response to added N is not surprising given that the surface 10-cm of soil contained measureable levels of $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$ at the end of the growing season (table 1), which indicates this soil contained adequate levels of N for sagebrush growth and microbial utilization.

Carbon substrate for heterotrophic decomposition appeared to be provided almost entirely by sagebrush leaf litter, which is very thin compared to appreciable and sometimes thick forest floor layers observed under deciduous and conifer forests. Carbon substrate and water limitations for heterotrophic respiration are probably common throughout shrublands so these results would be representative of other sagebrush stands under similar climatic conditions. During the dry season (mid-June through August in the Great Basin), rainfall events stimulate

pulses of soil respiration that characterize the highly skewed soil CO_2 spatial and temporal distributions found in our study. Thus, sufficient temporal measurements are needed to capture CO_2 respiration during and shortly after rainfall events. Infrequent measurements might result in underestimates of summertime soil CO_2 flux.

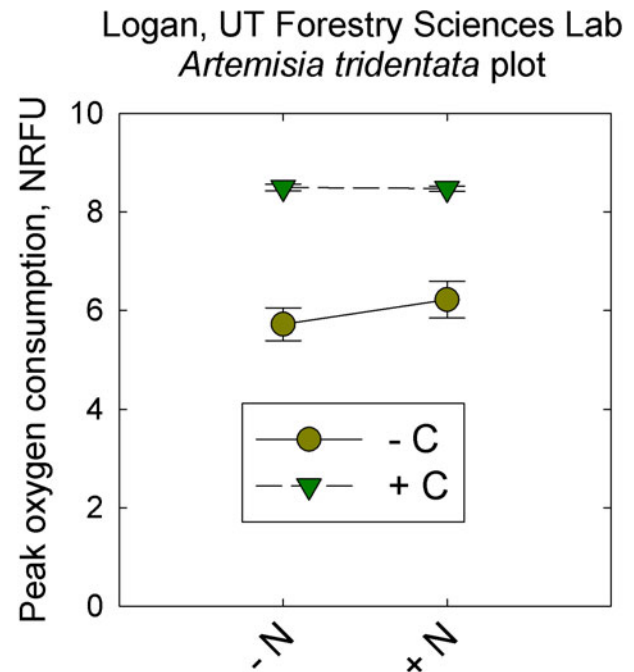


Figure 9. Effect of added N, C substrate (glucose), and N + C substrate on peak oxygen consumption by native microbe populations in a sagebrush stand at the Logan Forestry Sciences Laboratory.

Bahn and others (2010) showed that regardless of biome, total annual measured soil CO_2 flux is closely related to soil CO_2 flux measured at mean annual temperature (MAT). They based this finding on analysis of 80 site-years of soil respiration, temperature, and moisture data from 57 forests, plantations, savannas, shrublands, and grasslands from boreal to tropical climates. However, there were no sagebrush-dominant sites in their database. Their derived relationship is

$$\text{Annual soil CO}_2 \text{ flux} = 436.2 (\text{Soil CO}_2 \text{ flux at MAT})^{0.926}, r^2 = 0.94 (p < 0.001) \quad (\text{equation 1})$$

Mean annual temperature for our site from November 1, 2009 through October 31, 2010 was 10.59 ± 0.12 deg C. We used all the soil measurements collected when soil temperature was between 10.1 and 11.1 deg C to provide a larger pool of values (241

measurements) to calculate soil respiration at our MAT ± 0.5 deg C. We obtained a mean soil respiration value of $1.41 \pm 0.04 \text{ } \mu\text{mol m}^{-2} \text{ s}^{-1}$ at our MAT. This value is similar to calculated soil respiration values from other biomes with similar MAT values (Bahn and others 2010). This MAT soil respiration value predicts an annual total soil CO₂ flux of $598 \text{ g C m}^{-2} \text{ y}^{-1}$ using the Bahn and others (2010) equation 1 shown above. Actual annual total soil CO₂ flux calculated by summing all the measured and interpolated (for missing values) hourly values was $328 \text{ g C m}^{-2} \text{ s}^{-1}$ for our site, about half the predicted value from the Bahn and others (2010) equation.

However, for semi-arid and arid sites in which precipitation is less than potential evapotranspiration ($P < PET$), Bahn and others (2010) had to derive an aridity index based on P/PET to accurately estimate annual total CO₂ flux from soil respiration measured at MAT:

$$\text{Annual soil CO}_2 \text{ flux predicted/observed} = 1.278 - 0.601 \ln(P/PET), r^2 = 0.82 (p < 0.001) \quad (\text{equation 2})$$

Logan, Utah, is in a semi-arid region with a mean annual precipitation of 450 mm (1893-2005). Soil respiration at MAT tends to occur during the spring wet season. Throughout the summer months, precipitation is substantially less than ET so the predicted annual total soil CO₂ flux must be adjusted downward to account for suppression of soil respiration during dry periods when soil water content is very low. Bahn and others (2010) used PET to calculate their aridity index. However, PET is not easy to derive and depends on many factors. The Utah State University Climate Center reports daily precipitation and reference evapotranspiration (ET₀) values. Using that dataset we calculated a 30-year average (1980-2010) P/ET_0 for our site of 0.488 (504 mm/1033 mm). For the November 1, 2009 through October 31, 2010 period when we measured soil CO₂ flux, P/ET_0 was 0.396 (385 mm/972 mm), which was drier than the most recent 30-year period. Using a P/ET_0 of 0.396, the ratio of predicted to observed annual soil CO₂ flux from equation 2 is expected to be 1.83. The ratio of predicted (calculated from equation 1) to observed annual soil CO₂ flux was 1.82 ($598/328 \text{ g C m}^{-2} \text{ y}^{-1}$). Thus, equation 2 accurately calculated the correction factor needed to adjust predicted annual soil CO₂ flux to match the observed value for a semi-arid sagebrush site. Dividing the predicted annual soil CO₂ flux of $598 \text{ g C m}^{-2} \text{ y}^{-1}$ by the 1.83

correction factor calculated from equation 2, a corrected value of $327 \text{ g C m}^{-2} \text{ y}^{-1}$ is obtained for the predicted annual CO₂ flux, a number virtually identical to the observed result from summing the hourly measurements. Thus, annual soil respiration can be accurately calculated using 1) soil CO₂ measurements at or near MAT, 2) hourly soil temperature measurements throughout the year to calculate MAT, and 3) daily precipitation and ET₀ data from a nearby weather station.

Since soil temperature and water content are often routinely measured with soil respiration, it might be possible to develop a relationship between measured soil CO₂ flux and measured soil temperature and water content. Figures 3 and 4 reveal that the relationships among soil CO₂ flux, soil temperature, and soil moisture are complex. During dry summer months, soil temperature continues to increase, but soil respiration decreases as soil water content decreases. We found that the product of soil temperature and soil water content (soil temperature x soil water content), which we call the soil environment index (SEI), is linearly related to soil CO₂ flux (figure 10), although there is significant scatter. Only when soil temperature and water content are high does soil respiration reach its peak. Any combination of low temperatures or low soil moisture will tend to decrease soil respiration.

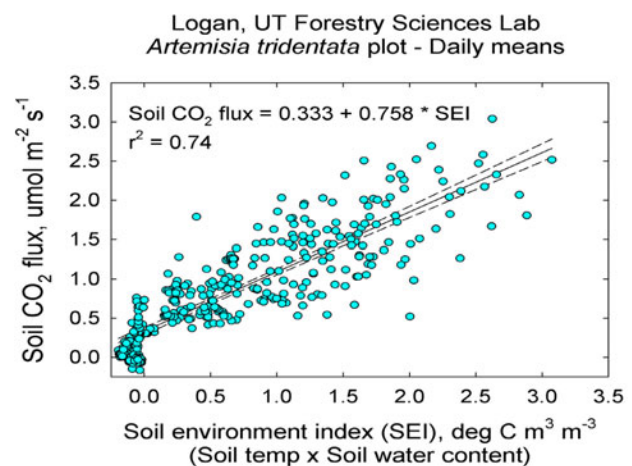


Figure 10. Relationship between soil CO₂ flux and soil environment index (defined as soil temperature x soil water content) for a sagebrush stand at the Logan Forestry Sciences Laboratory.

At times, large hourly variations in soil respiration in the sagebrush stand were observed. Although hourly soil respiration does not follow a predictable pattern,

monthly and seasonal values are strongly related to seasonal changes in soil temperature and soil water content and are therefore more predictable than short-term temporal changes. Once the seasonal relationship of soil CO₂ flux to soil temperature and water content is established for a given plant community and site combination (e.g., figure 10 for a sagebrush site), that relationship can be used to estimate soil CO₂ flux for areas or times with few soil respiration measurements provided soil temperature and moisture are measured frequently. We only have a single year of temporal variation in soil CO₂ flux at this site. Additional years of data are needed to determine variability across years as climatic conditions change from year to year.

The relatively small and fairly constant observed difference between daytime and nighttime mean soil respiration at this site indicates that it would be sufficient to characterize soil respiration at other sagebrush sites using daylight measurements with the survey chamber. Nighttime values could be modeled based on relative differences from more complete datasets at a limited number of sites. Because monthly and seasonal differences tend to be larger than daily differences, daytime survey type measurements at other sites during spring through fall months (e.g., weekly or even biweekly) may be sufficient to estimate seasonal soil CO₂ fluxes with the caveat that missing rainfall event effects on soil respiration would result in underestimation. Soil CO₂ flux can be estimated at times between survey measurements using soil temperature and moisture data collected hourly via dataloggers connected to the sensors and relationships such as that shown in figure 10.

An indication of the ability of monthly survey chamber (local spatial scale) measurements to represent monthly means from the long-term chamber can be seen in figure 11. Survey chamber soil CO₂ flux measurements at mid-month during the growing season overlapped with monthly means from the long-term chamber except in the spring (April and May) when monthly means were lower than survey-chamber means. Rapid changes in soil CO₂ flux occur at this time of year in response to large temperature and moisture changes. Thus, survey chamber measurements on a single day may differ substantially from monthly means measured with the long-term chamber. Soil temperatures measured during survey chamber soil CO₂ flux measurements

were always greater than monthly mean temperatures because monthly means includes lower nighttime soil temperature readings. Soil water content measured during survey chamber measurements were similar to monthly mean values since soil water content had little diurnal variation.

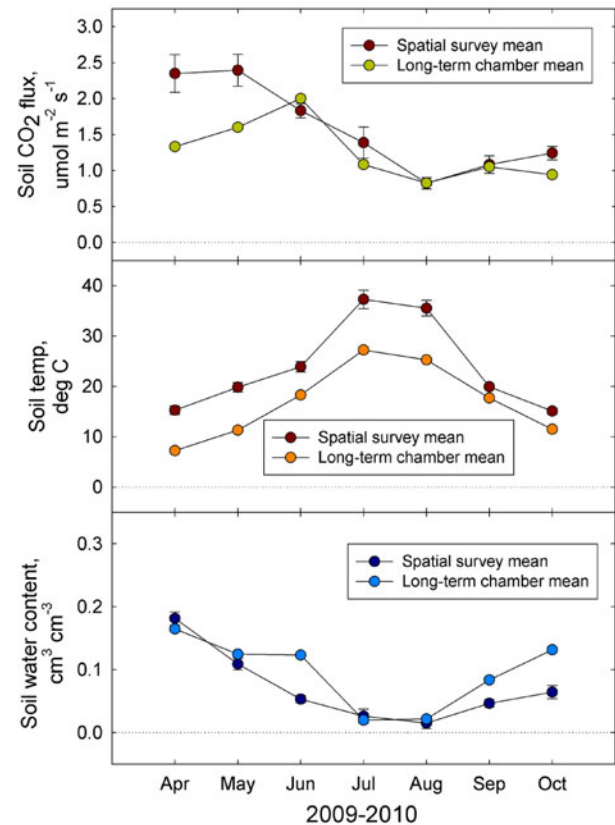


Figure 11. Monthly (April through October, 2010) survey (11 points) and long-term chamber (one point) mean $\pm$ std err soil CO₂ flux (top), temperature (middle), and water content (bottom) in a sagebrush stand at the Logan Forestry Sciences Laboratory.

Within site variability in soil respiration can be large (figure 9). Sufficient local-scale measurements are needed to fully characterize a given locale and reduce uncertainty about a site mean. Bradford and Ryan (2008) provided guidelines on the number of soil collars needed per site to adequately estimate local spatial variability to detect differences among sites due to plant community types, land use activities, or vegetation treatments that could influence soil respiration. Bradford and Ryan (2008) also provided guidelines for seasonal temporal sampling.

For landscape-scale estimates of soil CO₂ flux in shrublands in general or sagebrush-dominated lands in particular, small-scale spatial variance is of less

concern than having enough sites across the landscape to provide large-scale estimates of soil CO₂ fluxes. Unfortunately, landscape-scale estimates of soil CO₂ flux in sagebrush-dominated plant communities are largely unknown. Given the large spatial extent of sagebrush in the Great Basin and elsewhere, spatial variability of soil respiration under sagebrush is probably as great as or perhaps even greater than temporal variability as documented herein. A Forest Inventory and Analysis (FIA)-type sampling (Amacher and Perry 2010) of soil CO₂ flux across sagebrush-dominated plant communities in the Great Basin and other physiographic areas could provide the most accurate landscape-scale estimate. Stratification of sampling based on plant community types, landscape position, and topographic and geographic gradients (elevation, latitude, and longitude) would provide the most efficient sampling scheme to reduce uncertainties. A broad-scale survey would need to include plant communities with other sagebrush species (e.g., black sagebrush (*Artemisia nova* A. Nelson), plant communities in which sagebrush co-occurs with herbaceous species such as grasses (e.g., sagebrush-steppe ecosystems), and other types of shrublands (e.g., desert communities dominated by salt shrubs). Work is underway to try to determine large-scale spatial variability of soil CO₂ flux in a variety of plant community types across geographic gradients in the Interior West.

SUMMARY

Soil CO₂ flux, temperature, and water content exhibited large temporal and spatial variability in a sagebrush stand at the Logan FSL. Mean (std dev) soil CO₂ flux, temperature, and water content for the measurement period (November 1, 2009 - October 31, 2010) were 0.96 (0.81) $\mu\text{mol m}^{-2} \text{s}^{-1}$, 10.59 (10.11) deg C, and 0.101 (0.062) $\text{m}^3 \text{m}^{-3}$, respectively. Measured soil CO₂ flux within 0.5 deg of the mean annual soil temperature (10.6 deg C) averaged $1.41 \pm 0.04 \mu\text{mol m}^{-2} \text{s}^{-1}$. Using an equation from Bahn and others (2010) to predict annual soil CO₂ flux from measured soil CO₂ flux at MAT, our predicted annual soil CO₂ flux was $598 \text{ g C m}^{-2} \text{y}^{-1}$ for the sagebrush stand at the Logan FSL. The actual measured total obtained by summing all the hourly measurements was about half that ($328 \text{ g C m}^{-2} \text{y}^{-1}$). For semi-arid or arid sites where precipitation is less than evapotranspiration, measured total annual soil CO₂ flux will be less than the potential total because of dry season suppression of soil respiration when soil water content is very low. A correction factor based on local climate station P/ET₀ datasets correctly calculated

that the ratio of predicted to observed annual soil CO₂ flux was 1.83. Dividing the predicted annual soil CO₂ flux ($598 \text{ g C m}^{-2} \text{y}^{-1}$) by the correction factor (1.83) yielded an accurate prediction ($327 \text{ g C m}^{-2} \text{y}^{-1}$) of the measured annual soil respiration ($328 \text{ g C m}^{-2} \text{y}^{-1}$).

This study provides guidelines for capturing the temporal variability of soil CO₂ flux. Although our site is representative of other sagebrush sites under similar climatic conditions, a landscape-scale spatial survey is needed to estimate large-scale soil CO₂ flux for sagebrush-dominated landscapes.

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Selection and Vegetative Propagation of Native Woody Plants for Water-Wise Landscaping

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ABSTRACT

*Native woody plants with ornamental characteristics such as brilliant fall color, dwarf form, or glossy leaves have potential for use in water conserving urban landscapes. Individual accessions with one or more of these unique characteristics were identified based on the recommendations of a wide range of plant enthusiasts (both professional and amateur). Documentation of these accessions has been done through locating plants on-site where possible and then developing a record based on digital photography, GPS determined latitude and longitude, and place marking of Google Earth® images. Since desirable characteristics are often unique to a single plant, utilization of these plants by the landscape industry requires that they be clonally propagated. Methods of asexual propagation including grafting, budding, layering and cuttings may be successful with native plants, but are species and even accession specific. We report on the successful cutting propagation of *Arctostaphylos patula*, *A. pungens*, and *Cercocarpus intricatus*, and lack of success with *Juniperus osteosperma*, and *Mahonia fremontii*.*

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INTRODUCTION

There is a market for trees and shrubs native to the Intermountain west for use in low-water landscaping that conserves water without impacting landscape quality or function. Based on horticultural precedent, there is an even greater market value for exceptional clones of these native plants that not only conserve water, but bring aesthetic and functional value to the landscape. In order to take advantage of this market, it is important that highly ornamental accessions of native woody plants be identified and methods for their successful propagation and production be developed. Currently, improved selections of many of the native plants indigenous to the Intermountain area are not available in the nursery trade, and are therefore unavailable for water conserving landscapes.

We have documented over 32 species of native plants with one or more exceptional clones, and are currently investigating another 17. While clones of some, such as mountain lover (*Paxistima myrsinites*), can be easily propagated vegetatively, others have either never been tried or have shown only limited success. The purpose of our research is to select exceptional clones, determine optimum propagation methods, and make both the materials and the

methods available to the industry and the consuming public.

MATERIALS AND METHODS

Locating and Selecting Plants

The success of this project is a result of individuals willing to share their knowledge of unique specimens of native woody plants in Utah and adjacent states. We have polled botanists, natural resource managers, native plant enthusiasts and others regarding such plants, and are in the process of documenting suggested plants (tables 1 and 2). Some individual plants have been shown directly to us, while other suggestions have been referrals to general populations. In both cases we have found that in the process of documenting selected plants we have found additional plants with as good or even greater potential. Utilization of these plants in the landscape industry is dependent on their ease of propagation and production, and their performance in the landscape over an extended period of time. In reality, most of the plants listed will probably not be adopted for commercial production. But, some have great potential to enhance local landscaping and aid in water conservation.

Table 1. Native woody plants suggested for use in low-water landscaping, including the source of recommendation and the general location.

No.	Genus	Species	Source	Utah Counties or State
1	<i>Acer</i>	<i>glabrum</i>	Hale	Nevada
2	<i>Acer</i>	<i>glabrum</i>	Rupp	Sanpete
3	<i>Acer</i>	<i>glabrum</i>	Rupp	Sanpete
4	<i>Acer</i>	<i>glabrum</i>	Warner	Sevier
5	<i>Acer</i>	<i>grandidentatum</i>	Barker	Carbon
6	<i>Acer</i>	<i>grandidentatum</i>	Laub	Box Elder
7	<i>Acer</i>	<i>grandidentatum</i>	Morris	Cache
8	<i>Acer</i>	<i>grandidentatum</i>	Morris	Cache
9	<i>Acer</i>	<i>grandidentatum</i>	Morris	Cache
10	<i>Acer</i>	<i>grandidentatum</i>	Morris	Cache
11	<i>Acer</i>	<i>grandidentatum</i>	Morris	Cache
12	<i>Acer</i>	<i>grandidentatum</i>	Richards	Cache
13	<i>Acer</i>	<i>grandidentatum</i>	Richards	Cache
14	<i>Acer</i>	<i>grandidentatum</i>	Richards	Cache
15	<i>Acer</i>	<i>grandidentatum</i>	Richards	Cache
16	<i>Acer</i>	<i>grandidentatum</i>	Richards	Cache
17	<i>Acer</i>	<i>grandidentatum</i>	Richards	Cache
18	<i>Acer</i>	<i>grandidentatum</i>	Richards	Cache
19	<i>Acer</i>	<i>grandidentatum</i>	Richards	Cache
20	<i>Acer</i>	<i>grandidentatum</i>	Reid	Iron
21	<i>Acer</i>	<i>grandidentatum</i>	Reid	Iron
22	<i>Acer</i>	<i>grandidentatum</i>	Rupp	Cache
23	<i>Acer</i>	<i>grandidentatum</i>	Rupp	Cache
24	<i>Acer</i>	<i>grandidentatum</i>	Rupp	Cache
25	<i>Acer</i>	<i>grandidentatum</i>	Rupp	Cache
26	<i>Amelanchier</i>	spp.	Love	Bingham Co., ID
27	<i>Amelanchier</i>	spp.	Rupp	Rich
28	<i>Amelanchier</i>	<i>utahensis</i>	Bowns	Iron
29	<i>Amelanchier</i>	<i>utahensis</i>	Bowns	Iron
30	<i>Arctostaphylos</i>	<i>patula</i>	Rupp	Kane
31	<i>Arctostaphylos</i>	<i>patula</i>	Rupp	Garfield
32	<i>Arctostaphylos</i>	<i>patula</i>	Stevens	Sanpete
33	<i>Arctostaphylos</i>	<i>pungens</i>	Bowns	Washington
34	<i>Arctostaphylos</i>	<i>pungens</i>	Bowns	Washington
35	<i>Betula</i>	<i>occidentalis</i>	Rupp	Sevier
36	<i>Ceanothus</i>	<i>greggii</i>	Bowns	Washington
37	<i>Ceanothus</i>	<i>martinii</i>	Monsen	Sanpete
38	<i>Ceanothus</i>	<i>velutinus</i>	Rupp	Cache
39	<i>Ceanothus</i>	<i>velutinus</i>	Rupp	Rich
40	<i>Cercocarpus</i>	<i>intricatus</i>	Kjelgren	Cache
41	<i>Cercocarpus</i>	<i>intricatus</i>	Monsen	Beaver
42	<i>Cercocarpus</i>	<i>intricatus</i>	Rupp	Clark County, NV
43	<i>Cercocarpus</i>	<i>intricatus</i>	Rupp	Clark County, NV
44	<i>Cercocarpus</i>	<i>intricatus</i>	Stevens	Sanpete
45	<i>Cercocarpus</i>	<i>ledifolius (broom)</i>	Wildrick	Rich
46	<i>Cornus</i>	<i>sericea</i>	Rupp	Cache
47	<i>Ericameria</i>	<i>nauseosa</i> ssp.	Stevens	Juab
		<i>nauseosa</i> var. <i>speciosa</i>		
48	<i>Ericameria</i>	spp.	Anderson	Box Elder

Table 1 (cont.). Native woody plants suggested for use in low-water landscaping, including the source of

recommendation and the general location.

No.	Genus	Species	Source	Utah Counties or State
49	<i>Fraxinus</i>	<i>anomala</i>	Rupp	Emery
50	<i>Juniperus</i>	<i>osteosperma</i>	Rupp	Kane
51	<i>Juniperus</i>	<i>osteosperma</i>	Stevens	Sanpete
52	<i>Juniperus</i>	<i>scopulorum</i>	Rupp	Emery
53	<i>Juniperus</i>	<i>scopulorum</i>	Rupp	Emery
54	<i>Juniperus</i>	<i>scopulorum</i>	Stevens	Sanpete
55	<i>Juniperus</i>	<i>scopulorum</i>	Stevens	Sanpete
56	<i>Juniperus</i>	<i>x osteosperma</i>	Stevens	Sanpete
57	<i>Mahonia</i>	<i>fremontii</i>	Rupp	Cache
58	<i>Mahonia</i>	<i>fremontii</i>	Warner	Sevier
59	<i>Mahonia</i>	<i>repens</i>	Cope	Cache
60	<i>Mahonia</i>	<i>repens</i>	Cope	Cache
61	<i>Mahonia</i>	<i>repens</i>	Cope	Cache
62	<i>Mahonia</i>	<i>repens</i>	Rupp	Sanpete
63	<i>Paxistima</i>	<i>myrsinites</i>	Rupp	Teton County, WY
64	<i>Philadelphus</i>	<i>microphyllus</i>	Rupp	Emery
65	<i>Pinus</i>	<i>edulis</i>	Stevens	Sanpete
66	<i>Pinus</i>	<i>edulis</i>	Stevens	Sanpete
67	<i>Purshia</i>	<i>tridentata</i>	Rupp	Millard
68	<i>Quercus</i>	<i>gambelii</i>	Rupp	Millard
69	<i>Quercus</i>	<i>gambelii</i>	Rupp	Beaver
70	<i>Quercus</i>	<i>gambelii</i>	Stevens	Sanpete
71	<i>Quercus</i>	<i>pauciloba</i>	Bowns	Iron
72	<i>Rhus</i>	<i>aromatica simplicifolia</i>	Bowns	Washington
73	<i>Rhus</i>	<i>glabra cismontana</i>	Stevens	Juab
74	<i>Salvia</i>	<i>dorii 'Clokeyi'</i>	Anderson	Clark County, NV
75	<i>Shepherdia</i>	<i>rotundifolia</i>	Rupp	Washington

Full names of sources include: Richard Anderson, Philip Barker, James Bowns, Kevin Cope, Eric Hale, Roger Kjellgren, Thomas Laub, Steven Love, Stephn Monsen, Jerry Morris, Chad Reid, Melody Richards, Larry Rupp, Richard Stevens, Janett Warner, and Carl Wildrick.

Documenting Plants and Locations

Current technology has made the documentation of individual plants a simple process. Identified accessions are documented with digital photography and the latitude and longitude determined by GPS (Garmin GPSMAP®60CS or 60CSx). We have also found it helpful to place-mark the accession on a Google Earth® image to facilitate finding it (figure 1).

Vegetative Propagation

In horticultural production systems, asexual propagation of clonal material is used to establish the large numbers of uniform plants demanded by the industry and the consuming public. The characteristic of genetic diversity within a selected population of plants so desirable in reclamation is not a priority,

since the high value of horticultural crops allows economic management of the problems that occur with clonal populations. Our goal with vegetative propagation has two parts. First, we are interested in asexual propagation as a means of initially establishing clones of wild plant materials in a nursery environment (figures 2 and 3). Once established, we are then focusing on how to economically propagate large numbers of the selected clone in a nursery setting. Given that rooting of cuttings is a genetic trait, determining the best method of propagation is not trivial. Response to cuttings or other propagation methods can vary significantly between clones. Research done with nursery-grown stock plants is also much more applicable to commercial production nurseries and will help us in our goal of assisting nurserymen of the interior western states to produce these plants.

Table 2. Native woody plants suggested for use in low-water landscaping, but not yet fully documented, including the source of recommendation and the documented location.

No.	Genus	Species	Source	Utah Counties or State
1	<i>Amelanchier</i>	<i>alnifolia</i>	Monsen	Iron
2	<i>Amelanchier</i>	<i>utahensis</i>	Monsen	Washington
3	<i>Arctostaphylos</i>	<i>patula</i>	Monsen	Wayne
4	<i>Artemisia</i>	<i>cana</i>	Monsen	Bighorn County, .MT
5	<i>Artemisia</i>	<i>cana</i>	Schulz	Eastern Wyoming
6	<i>Artemisia</i>	<i>filifolia</i>	Bowns	Washington
7	<i>Artemisia</i>	<i>nova</i>	Monsen	Juab
8	<i>Artemisia</i>	<i>pedatifida</i>	Schulz	Eastern Wyoming
9	<i>Artemisia</i>	<i>rigida</i>	Monsen	Ada County, ID
10	<i>Artemisia</i>	<i>rigida</i>	Schulz	Washington State
11	<i>Artemisia</i>	<i>rothrockii</i>	Monsen	Sanpete
12	<i>Atriplex</i>	<i>hymenelytra</i>	Stevens	Washington
13	<i>Ceanothus</i>	<i>prostratus</i>	Hanson	Adams County, ID
14	<i>Ceanothus</i>	<i>prostratus</i>	Monsen	McCall County, ID
15	<i>Ceanothus</i>	<i>prostratus</i>	Monsen	Sanpete
16	<i>Ceanothus</i>	<i>prostratus</i>	Monsen	Ada County, ID
17	<i>Ceanothus</i>	<i>velutinus</i>	Monsen	Ada County, ID
18	<i>Cercocarpus</i>	hybrid	Monsen	Sheridan County, WY
19	<i>Cercocarpus</i>	<i>intricatus</i>	Monsen	Millard
20	<i>Cercocarpus</i>	<i>intricatus</i>	Monsen	White Pine County, NV
21	<i>Cercocarpus</i>	<i>intricatus</i>	Monsen	Rio Blanco County, CO
22	<i>Cercocarpus</i>	<i>ledifolius</i>	Schulz	Juab
23	<i>Cercocarpus</i>	<i>ledifolius</i> x <i>montanus</i>	Monsen	Sheridan County, WY
24	<i>Cupressus</i>	<i>arizonica</i>	Monsen	Sanpete
25	<i>Ericameria</i>	<i>nauseosus salicifolia</i>	McArthur	Sanpete
26	<i>Fallugia</i>	<i>paradoxa</i>	Monsen	Sevier
27	<i>Juniperus</i>	<i>scopulorum</i>	Monsen	Bighorn County, .MT
28	<i>Juniperus</i>	<i>scopulorum</i>	Monsen	Caribou County, ID
29	<i>Juniperus</i>	<i>scopulorum</i>	Stevens	Sanpete
30	<i>Peraphyllum</i>	<i>ramosissimum</i>	Hanson	Washington County, ID
31	<i>Physocarpus</i>	<i>alternans</i>	Kitchen	Millard
32	<i>Pinus</i>	<i>edulis</i>	Stevens	Millard
33	<i>Populus</i>	<i>tremuloides</i>	Stevens	Sanpete
34	<i>Populus</i>	<i>Tremuloides</i>	Reid	Iron
35	<i>Prunus</i>	<i>virginiana</i>	Welsh	Utah
36	<i>Purshia</i>	<i>tridentata</i>	Monsen	Sanpete
37	<i>Quercus</i>	<i>gambelii</i>	Stevens	Sanpete
38	<i>Quercus</i>	<i>turbinella</i>	McArthur	Salt Lake
39	<i>Salvia</i>	<i>argentea</i>	Schulz	Eastern Wyoming
40	<i>Symphoricarpos</i>	<i>Longiflorus</i>	Kitchen	Millard

Full names of sources include: James Bowns, Alma Hanson, Stanley Kitchen, Durant McArthur, Stephen Monsen, Leila Shultz, Richard Stevens, Janett Warner, and Stanley Welsh.

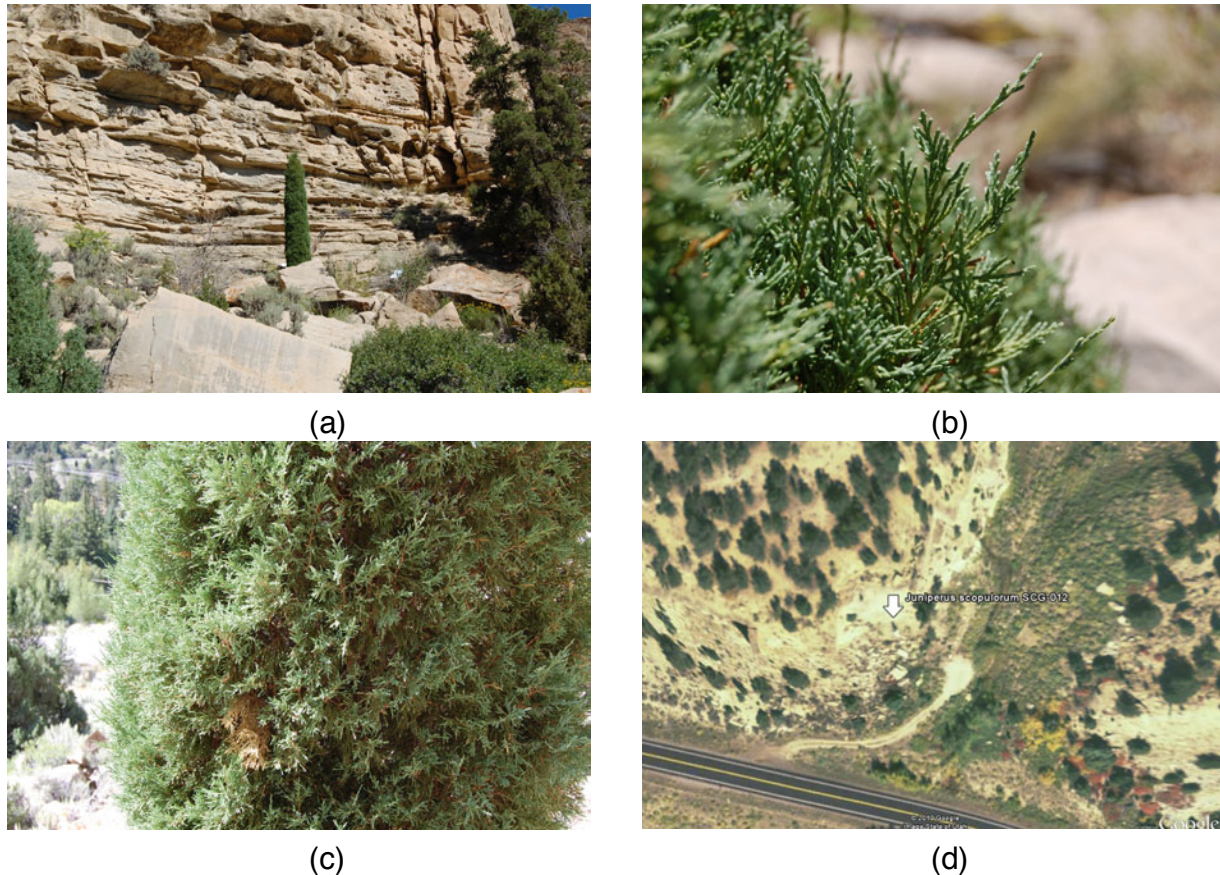


Figure 1. Documentation of an exceptionally columnar form of *Juniperus scopulorum*; site view with plant (A), close up of foliage (B), mid-view of foliage (C), and location on Google Earth[®] image (D).

Propagation of Evergreen Shrubs by Hardwood Cuttings

In an effort to further define vegetative propagation requirements of native shrubs, cuttings of previous season's growth were collected on Jan. 19-21, 2010 from a number of native shrubs (table 3) and propagated in a glass greenhouse in Logan, Utah. Cuttings were initially held on ice in a portable cooler until placed in a refrigerated storage at 4° C until Jan. 22-23 when stuck in a 4 perlite : 1 sphagnum peat (by volume) rooting substrate, with a reverse osmosis water mist (7 s/30 min during light period) and approximately 22-28° C bottom heat in a 18/16° C day/night greenhouse and 18 hour day length (using high pressure sodium lamps). The effect of auxin on rooting was examined by treatments of 0/0, 2000/1000, and 4000/2000 ppm indolebutyric acid (IBA)/naphthaleneacetic acid (NAA) as Dip 'N Grow[®] (Clackamas, Oregon) diluted in a 50 percent ethanol solution applied as a 5 s quick dip (n=12). Cutting positions were periodically randomized on the mist

bench. Rooting was evaluated after 7 weeks for all plants except juniper which was evaluated after 15 weeks. Evaluations consisted of determining the percentage of rooted cuttings and the number of roots per rooted cutting (root primordia were classified as roots if their length exceeded their width).

Statistical analysis of the percentage of rooted cuttings was done with logistic regression since the data have a binomial distribution and the method calculates a standard error value independent of cutting performance. Because the number of roots per cutting is considered count data, that analysis was done by ANOVA using square-root transformed data (Compton 2008).

RESULTS AND DISCUSSION

The effect of auxins on rooting as determined by percentage of rooted cuttings and number of roots per cutting showed a great deal of intra- and inter-specific variability (tables 4 and 5).

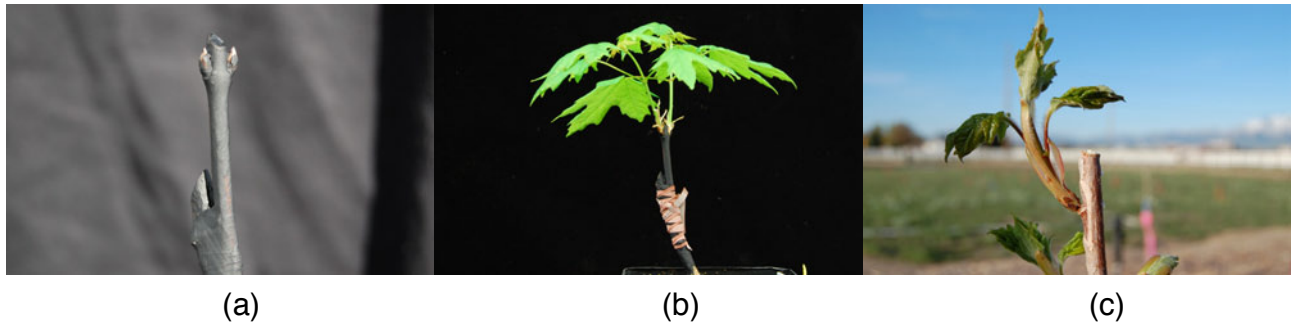


Figure 2. Side-veneer grafts (A & B) and chip budding (C) of *Acer grandidentatum* scions on seedling rootstocks as a means of clonal propagation and of establishing wild plant material in a controlled nursery environment.



Figure 3. Asexual propagation of cuttings using intermittent mist with bottom heat (A) and container packs (6 cm L x 5.5 cm W x 7 cm H) with 4:1 perlite:peat rooting medium (B). Successful propagation of hardwood (dormant) cuttings of *Arctostaphylos patula* (C) and *Cercocarpus intricatus* (D), and of semi-hardwood cuttings of *Ericameria nauseosa* ssp. *nauseosa* var. *speciosa* (E).

Table 3. Shrub sources for hardwood cuttings used in propagation trials.

Accession	Source	Notes
<i>Arctostaphylos patula</i> (002)	Sanpete County, Utah	High elevation
<i>Arctostaphylos patula</i> (014)	Kane County, Utah	Easily rooted
<i>Arctostaphylos pungens</i> (020)	Washington County, Utah	Large, multiple stems
<i>Arctostaphylos pungens</i> (021)	Washington County, Utah	Single stem form
<i>Cercocarpus intricatus</i> (003)	Sanpete County, Utah	Columnar form
<i>Juniperus osteosperma</i> (005)	Sanpete County, Utah	Purported hybrid, deer resistant
<i>Mahonia fremontii</i> (016)	Sevier County, Utah	Wildland Nursery stock block

Table 4. The effect of 0/0, 2000/1000, and 4000/2000 ppm indolebutyric acid (IBA)/naphthaleneacetic acid (NAA) as Dip 'N Grow[®] on percentage of rooted cuttings of selected specimens of native shrubs (n=12). Plant abbreviations are: *Arctostaphylos patula* (002) [ArcPat 002], *A. patula* (014) [ArcPat 014], *A. pungens* (020) [ArcPun 020], *A. pungens* (021) [ArcPun 021], *Cercocarpus intricatus* (003) [CerInt 003], *Juniperus osteosperma* (005) [JunOst 005], and *Mahonia fremontii* (016) [MahFre 016].

	ArcPat 002	ArcPat 014	ArcPun 020	ArcPun 021	CerInt 003	JunOst 005	MahFre 016
Rooting Hormone	Percentage of Rooted Cuttings						
0 ppm IBA/NAA	67	0	17*	67	0*	0	0
2000/1000 ppm IBA/NAA	92	0	83*	42	33*	0	0
4000/2000 ppm IBA/NAA	92	0	83*	58	42*	0	0

*Columns with asterisked data indicate a significant effect of rooting hormone on the percentage of rooted cuttings as shown by logistic regression at P=0.05 as calculated with Statistix 9 (Analytical Software 2008). Values of 0 were analyzed as 0.000001 for CerInt 003.

Table 5. The effect of 0/0, 2000/1000, and 4000/2000 ppm indolebutyric acid (IBA)/naphthaleneacetic acid (NAA) as Dip 'N Grow[®] on roots per rooted cutting of selected specimens of native shrubs (n=12). Plant abbreviations are: *Arctostaphylos patula* (002) [ArcPat 002], *A. patula* (014) [ArcPat 014], *A. pungens* (020) [ArcPun 020], *A. pungens* (021) [ArcPun 021], *Cercocarpus intricatus* (003) [CerInt 003], *Juniperus osteosperma* (005) [JunOst 005], and *Mahonia fremontii* (016) [MahFre 016].

	ArcPat 002	ArcPat 014	ArcPun 020	ArcPun 021	CerInt 003	JunOst 005	MahFre 016
Rooting Hormone	Average Number of Roots per Rooted Cutting						
0 ppm IBA/NAA	4.8 a ¹	0	4.0 a	6.6 a	0 a	0	0
2000/1000 ppm IBA/NAA	11.8 b	0	10.3 a	16.8 b	3.0 b	0	0
4000/2000 ppm IBA/NAA	8.3 ab	0	17.9 b	6.0 a	7.4 b	0	0

¹Means followed by different letters within columns are significantly different based on Analysis of Variance of square-root transformed data at P=0.05 and pairwise comparisons using Least Significant Differences completed with Statistix 9 (Analytical Software 2008). Values of 0 were analyzed as 0.000001 for CerInt 003.

Arctostaphylos

Cuttings of wild manzanita (*Arctostaphylos*) generally root better when taken as terminal cuttings during the winter, though cultivated plants can be more readily rooted year round (Borland and Bone 2007; Trindle and Flessner 2002). Our results were similar with good numbers of roots and rooting percentages of up to 92 percent of selection *A. patula* (002) (greenleaf manzanita) when treated with supplemental auxins. While it failed in this experiment, we have successfully rooted *A. patula* (014) previously (Rupp 2009, unpublished data) and it was noted that this group of cuttings had symptoms indicative of stem rot. Borland and Bone comment on the prevalence of *Phytophthora* (root rot) as a significant and generally fatal disease of *Arctostaphylos* cuttings, suggesting that greater attention to sanitation and the use of fungicides may be of benefit.

Cercocarpus intricatus

A review of the literature has shown no record of propagation for littleleaf mountain-mahogany (*C. intricatus*) by cuttings. Our research allows this method to be successful and that there is a significant effect of auxin treatments on both the percentage of rooted cuttings and the number of roots per cutting. While we were only able to root 42 percent of the cuttings taken, the number is high enough to suggest that fine-tuning the propagation process should increase rooting to a commercially acceptable level.

Juniperus osteosperma

Junipers are a very commonly used plant in the landscape horticulture industry with multiple references regarding their propagation (Dirr and Heuser 2006; Hartmann and others 2011). In general upright selections of the genus *Juniperus* are considered difficult to propagate by cuttings (Connor 1985). Vegetative propagation of Utah juniper (*J. osteosperma*) has been studied very little, with only one citation of success in the literature (Reinsvold 1986). In this study we attempted to propagate a purported hybrid of Utah juniper found in Sanpete County, Utah. While the mother plant has desirable characteristics in both form and deer resistance, we were unable to induce any root formation, even when extending the rooting time to 15 weeks. Successful propagation of this accession may require the use of grafting to establish it in a nursery environment followed by empirical applications of treatments such as length of propagation time, wounding, rooting hormone formulations, rejuvenation, and others.

Mahonia fremontii

Similarly to junipers, there are a number of species within *Mahonia* that are used in the landscape industry – including the native Frémont's mahonia (*M. repens*). Cuttings of these plants can be successfully rooted, though the ease of rooting varies with the species and cultivar. Propagation of *M. fremontii* by vegetative means has not been recorded in the literature. A preliminary study of rooting cuttings of *M. fremontii* showed successful rooting (Rupp 2010, unpublished data). However, in this experiment there was no rooting, but rather a blackening of the cutting stem bases. Based on research with other *Mahonia* species, the blackening and lack of rooting could be due to the time of year the cuttings were taken (Dirr and Heuser 2006). It is also interesting to note that all methods of *Mahonia* propagation reviewed in Dirr and Heuser used talc as the rooting hormone carrier, and our preliminary experiment also successfully used a talc carrier, which raises a question as to the suitability of the alcohol-based quick dip used in this experiment.

CONCLUSIONS

The potential for selecting exceptional specimens of native woody plants for use in water-conserving landscapes is very good and we have successfully identified a number of plants with potential for use in the industry. Asexual propagation to preserve genotypes is also successful in many cases. In those cases where clones from genera known to form adventitious roots (in other words *Juniperus* and *Mahonia*) did not root, further research is required to determine if these selections are genetically recalcitrant or if factors such as disease, timing, conditions when collecting, and/or storage practices are inhibiting rooting. Both improved propagation techniques and observation of selections over several years in a landscaped environment are required before these plants can be promoted for use in the industry. We continue to search for plants with the drought, cold, and soil hardiness needed for the intermountain area and the aesthetic attributes that would contribute to residential landscapes.

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Trend of Gardner Saltbush and Halogeton in the Lower Green River Basin, Wyoming

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ABSTRACT

*Displacement of Gardner saltbush (*Atriplex gardneri*) by halogeton (*Halogeton glomeratus*) is being recorded at several points in the Lower Green River Basin, Wyoming by line intercept measurements and by repeat photography. This paper gives results of the monitoring studies as of 2009. Total displacement of Gardner saltbush by halogeton has taken as little as 10 years at some locations. Loss of Gardner saltbush to halogeton has major management implications.*

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INTRODUCTION

Gardner saltbush (*Atriplex gardneri*) is a valuable resource. It forms relatively high producing stands on clay soils in areas of 16-20 cm (6.3-7.9 in) annual precipitation (Fisser et al. 1974) where few other plants have capacity to grow and even fewer have capacity to produce biomass anywhere near that of Gardner saltbush. In favorable years leaves of Gardner saltbush remain green through the winter, providing high quality forage for wildlife and livestock in the winter season. Although protein content of Gardner saltbush is comparatively low for a shrub (Cook and others 1954; Krysl and others. 1984), the winter-green nature of the plant indicates comparatively high levels of protein in winter when grasses and forbs are dry. Gardner saltbush is capable of persisting under levels of use as high as 35-50 percent (Blaisdell and Holmgren 1984; Cook 1971; Fisser and Joyce 1984). Heavy use of this and other desert shrubs can be expected to be detrimental (Cook and Stoddard 1963), and thus facilitate displacement of shrubs by weedy species such as halogeton (*Halogeton glomeratus*).

Halogeton is a highly invasive species of Eurasian origin that was first detected in North America near Wells, Nevada in 1934 (Dayton 1951; Holmgren and Anderson 1970; Young 2002) from where it spread across much of the arid and semiarid West via roadways and other disturbance corridors. It was unknown in Utah until about 1942 (Stoddard and others. 1949; Stoddard and others. 1951). Area of infestation as of 1952 was estimated at 1.5 million acres with 250,000 of these acres in Idaho (Tisdale

and Zappettini 1953). By 1954 it had spread across the deserts of the Great Basin, Colorado Basin of Utah and Colorado, and Wyoming. In the 1980's it continued to spread in its previously documented range and to new areas in Nebraska, Montana, Oregon, New Mexico and California. Pemberton (1981) expected additional advance in the Great Plains, eastern half of Washington, Arizona, and other areas. The PLANTS Data Base (USDA, NRCS 2010) currently shows halogeton in all of the 11 western states and South Dakota and Nebraska, and in all counties of Utah.

Halogeton produces both black and brown seeds. The black seeds germinate readily whenever moisture and temperatures are favorable, and they comprise about 66 percent of the seed crop (Cronin and Williams 1966). The black seeds are viable for only about 1 year, but they provide a means for rapid expansion once halogeton invades a suitable site. Brown seeds are dormant at dispersal, with only a small percentage germinating each year. Brown seeds remain viable in the soil for at least 10 years and thus provide a means for halogeton survival during long periods of drought (Cronin and Williams 1966).

Halogeton can change soil chemistry and soil ecology (Duda and others 2003) by means of salt pumping (Eckert and Kinsinger 1960) which reduces establishment of other plants (Kinsinger and Eckert 1961). Aqueous extracts of halogeton tissue can greatly reduce germination and growth of seedlings of other species (Smith and Rauchfuss 1958). The soil altering capability of halogeton is likely a major factor in the die-off of Gardner saltbush reported in this paper.

With the combination of abundant production of both kinds of seeds that together provide for both rapid spread and persistence in times of drought, and the capability to induce toxic soil conditions (Duda and others 2003; Eckert and Kinsinger 1960) halogeton is well equipped to persist as a dominate.

The delayed germination of brown seeds can be expected to make control of this plant difficult. Treatments that provide control for a year or two can be expected to be overrun by halogeton in a few years due to new recruitment from the seed bank. Tisdale & Zappettini (1953) found 80 percent of halogeton germination was in March, April, and May, with 16 percent in June, 2 percent in July, and 2 percent in August. This indicates a single application of herbicides in one year will not control halogeton. However, application of herbicides have facilitated establishment of perennial grasses in halogeton infested areas (Cook 1965; Hass and others 1962; Miller 1956).

Bleak and others (1965) made observations of 107 separate plantings in the shadscale [*Atriplex confertifolia* (Torrey & Fremont) Watson] zone in which 148 selected species of grasses, forbs, or shrubs were planted from 1937 to 1962. They concluded that these seedlings usually failed. A few exceptions were found where crested wheatgrass, Siberian wheatgrass, and Russian wildrye were planted. Hull (1963) reported results of seeding various native and introduced grasses into salt-desert shrub communities in 1948 and 1949. Of the plants included in his paper, Russian wildrye (*Elymus junceus* Fischer) was generally the most successful. However, this species as well as others failed or produced scattered stands by 1960. The papers by Bleak (1965) and Hull (1963) suggest difficulty in displacing halogeton with perennial species in the desert shrub zone.

Halogeton contains poisonous oxalates which has caused extensive losses especially when hungry sheep were trailed or transported to heavily infested areas (Kingsbury 1964). Most of a band of 1,600 sheep were lost over a 3-day period to this plant in the Raft River Valley of southern Idaho in 1945 when sheep were trailed from mountains into a halogeton infested range (Sharp and others 1990). Poisoning of cattle has also been reported (Bruner and Robertson 1963). Soluble oxalate content can be as high as 28 percent in the early fall. By spring oxalates drop to as low as 5 percent unless plants are covered by snow

(Cook and Stoddart 1953). The plants retain enough oxalates when dry in winter to remain toxic to livestock (Cook and Gates 1960).

STUDY AREA

Locations of study sites for this paper are within the Flaming Gorge National Recreation Area, Ashley National Forest and adjacent to this area in the lower Green River Basin, Sweetwater County, Wyoming. Studies files are kept at Ashley National Forest, Flaming Gorge District at Manila, Utah and at the Ashley National Forest Supervisors Office, Vernal, Utah. These studies are also filed electronically on external hard drives. They are numbered and filed in a geographic system using 7.5 minute US Geological Survey quadrangle maps as a basis.

Average annual precipitation at the Black Mountain Exclosure (41°15'40.8" North and 109°37'03.0" West) near Buckboard Crossing is 21.8 cm (8.17 in). Annual precipitation for Gardner saltbush communities of the area might be slightly less than at the Black Mountain Exclosure.

The general area supports mixed desert shrub communities of shadscale, winter fat [*Krascheninnikovia lanata* (Pursh) Meeuse & Smit], bud sagebrush (*Artemisia spinescens* D. C. Eaton), and other shrubs. Wyoming big sagebrush [*Artemisia tridentata* var. *wyomingensis* (Beetle & A. Young) Welsh] communities are found on uplands, and spiny hopsage [*Grayia spinosa* (Hooker) Moquin in DC.] communities are on aeolian sand and slope wash colluvium. Gardner saltbush communities are confined to clay soils which are generally found on flats at lower elevations of the area. The study area is underlain by the Green River Formation. However, it appears that some of the Gardner Saltbush sites are influenced by materials eroded from the adjacent Bridger Formation.

Halogeton was found in the area in 1973 in a rangeland survey of the Flaming Gorge National Recreation Area at study site 75-4 (numbering system explained above). However, this invasive species did not seem to greatly impact ecology of the area until robust growth of the species followed favorable precipitation in 2003 which in turn followed the severe drought of 2002. Gardner saltbush was greatly impacted by the drought of 2002. In 2003 halogeton plants of 45 cm (18 in) tall grew in great abundance in Gardner saltbush communities.

Wyoming Highway 530 and numerous roads in the Flaming Gorge National Recreation Area are most probable vectors of spread of halogeton in the study area. Reconstruction of Highway 530 in 2005 was particularly favorable for abundant growth of this weedy plant. The fluctuating water level of Flaming Gorge Reservoir is also a major contributor to the abundance of this plant. In some low-gradient areas, the draw-down basin of the reservoir supports nearly solid stands of halogeton. In desert shrub communities, of this study area halogeton is often seen first on prairie dog mounds.

METHODS

Long-term monitoring studies were established at several sites in Gardner saltbush communities. Trend of Gardner saltbush was determined by line intercept measurements of crown cover on eight permanently marked transects and by repeat photography of the same transects from permanently marked camera points. Line intercept measurements were taken along five 100 ft (30.5 m) transects for a total of 500 ft (152 m) at each of the eight study sites. Center points for study sites were arranged so that all of the belt lines were included in within Gardner saltbush communities. Sites were spaced across much of the Gardner saltbush type within the Flaming Gorge National Recreation Area. Repeat photography was also taken at some sites without Line intercept measurements. Nested frequency studies (USDA, Forest Service 1993) were initiated at some of the study sites. However, it became evident that nested frequency information was not needed to determine trend of Gardner Saltbush. Also frequency provides a poor expression of the high variability of volume or production of halogeton. This plant can have high frequency in years of low production as well as in years of high production. It is apparent that volume of herbage produce by halogeton had much more to do with community dynamics than did frequency of halogeton. Line intercept measurements gave direct and easily understood trend of Gardner saltbush. For these reasons nested frequency studies were not repeated.

RESULTS AND DISCUSSION

Table 1 shows trend in percent crown cover of Gardner saltbush for the eight study sites. Crown cover measurements taken in 2009 indicate a downward trend for Gardner saltbush at all eight sites.

This trend is most obvious at study sites 72-13 (figures 1, 2, 3, 4 and 5), 72-18, and 72-31B. These studies experienced total or near total conversion of Gardner saltbush communities to halogeton within a 15-year interval. It is likely that sites with greater concentrations of salts are more rapidly converted.

Study site 6-16 demonstrates a wide shift in crown cover of Gardner saltbush from year to year with a high of 31.5 percent in 1992 and a low of 4.4 percent in 2003. The low reading of 2003 followed the drought of 2002. One year later in 2004 crown cover returned to nearly the level of 1992. However, in the highly favorable moisture year of 2005, there was a decline of about 4 percent. A continuing decline is reflected by the 19.7 percent crown cover measured in 2009. The wide variation in cover from year to year indicates a need for frequent monitoring. Table 1 reveals a greater frequency of monitoring after 2003. This greater frequency was prompted by recognition of this yearly variation.

Study site 75-7 shows increased crown cover of Gardner saltbush from 1991-2005. This increase was associated with exclusion of livestock at this study which was fenced in 1991. However, the 2009 measurement indicates a trend of decreasing crown cover even in the absence of livestock. Photography of the site demonstrates an increase in the area dominated by halogeton within the exclosure. This study indicates livestock grazing might contribute to decrease in Gardner saltbush cover. However, it also indicates conversion to halogeton will take place in absence of livestock.

Study site 72-13 (figures 1, 2, 3, 4, and 5) experienced total loss of Gardner saltbush cover over a 16 year-interval with a high of 25.8 percent crown cover in 1993 and a low of 0 percent in 2009. The reading of (2.1 percent) in 2005 (a year of highly favorable precipitation) suggests strongly that Gardner saltbush will not recover at this site. Repeat photography as well as line intercept measurements at this site demonstrate that halogeton has displaced Gardner saltbush. Study site 72-18 with crown cover of Gardner saltbush at 23.6 percent in 1998 and 4.9 percent in 2009 indicates the trend seen at site 72-13 where crown cover of this shrub was zero in 2009.

Table 1. Trend in percent crown cover* of Gardner saltbush at 8 sites.

Study	Year													
	1989	1991	1992	1993	1994	1997	1998	1999	2003	2004	2005	2007	2008	2009
6-16			31.5						4.4	28.2	24.8			19.7
72-4	7.5										8.5			4.8
72-13				25.8							2.1			0.0
72-18							23.6		3.4	7.4	13.0	7.2		4.9
72-31B								13.4			0.4			0.0
75-7		3.6			7.4	8.1					8.8			3.4
75-8B											6.3			1.2
75-21B								13.2		4.3	10.0	5.1		5.7

*Values for crown cover are based on measurements taken along 500 ft (152 m) of line intercept at each of the above sites.

Line intercept studies with 200 ft (60 m) of intercept were established in 2009 at study sites 71-11C, 71-11D, and 71-11E. These studies are within 200 ft (60 m) of each other, and they are similar in gradient, aspect, and apparent soil features. Crown cover of Gardner saltbush was 1.5 percent, 27.4 percent and 3.4 percent at these sites respectively. The wide variation in Gardner saltbush cover at these similar and closely spaced sites appeared to be related to past abundance of halogeton as detected by remnant plants.

In addition to Line Intercept, repeat photography of study sites also demonstrated the decline and displacement of Gardner saltbush. Study site 75-4 was photographed in 1991, 1995, 1996, 1998, 2005, 2007, 2008 and 2009. Based on comparison of photos of other sites with line intercept measurements, the 1990's photos of this site indicate Gardner saltbush crown cover at 15 percent-20 percent with little cover of halogeton. Photos of 2005 show high percent cover of halogeton with little Gardner saltbush. In this case conversion occurred during a 7-year interval between 1998 and 2005. The photo record at this site vividly demonstrates rapid conversion following the drought of 2002-2003.

The photo record of study site 72-31A demonstrates conversion from an estimated 25 percent crown cover of Gardner saltbush in 1999 to an estimated 3 percent in 2005 and near 0 percent in 2009. Photos and measurements at some sites taken in 2009 including site 71-11D show Gardner saltbush persisting with high levels of crown cover. These sites have low cover of halogeton.

Repeat photography at the study sites also shows boom and bust dynamics of halogeton with abundant growth in some years and essentially no growth in

others. Years of little growth of halogeton followed by years of abundant growth indicate that the halogeton seed-bank persists in the soil (Cronin and Williams 1966). This seed-bank has the potential to release in years of favorable precipitation.



Figure 1. 24 May 1993. Crown cover of Gardner saltbush is 37.6% on this beltline with an average of 25.8% for 5 beltlines. This condition follows a century of livestock grazing.

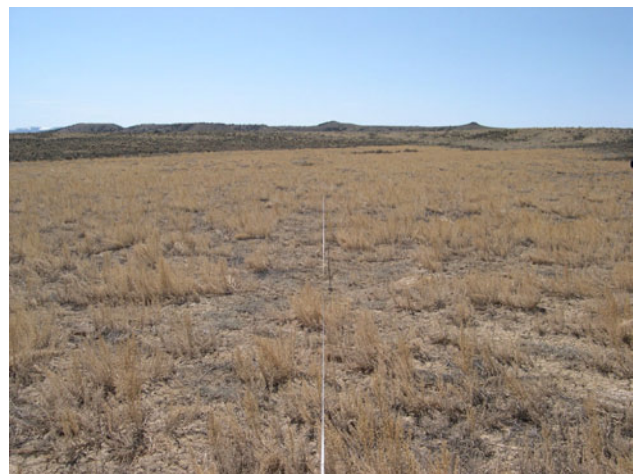


Figure 2. 31 March 2004. Large skeletons of halogeton indicate abundant growth of this annual in 2003.



Figure 3. 3 June 2005. Following high production of halogeton in 2003, crown cover of Gardner saltbush is 1.7% on this beltline with an average of 2.1% for 5 beltlines.

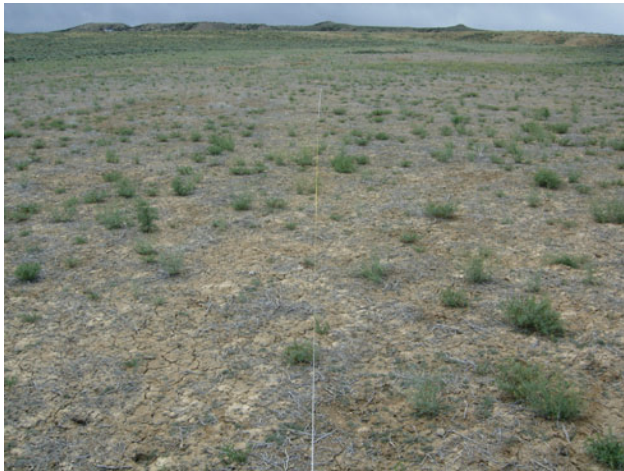


Figure 4. 5 May 2007. Crown cover of Gardner saltbush was not measured in 2007. However, it appears to be slightly higher than in 2005. Essentially all green vegetation in the area of the beltline is Gardner saltbush. The gray litter on the ground is dried halogeton.

MANAGEMENT IMPLICATIONS

Gardner saltbush persisted through the 1990's with comparatively light growth of halogeton. Gardner saltbush currently persists as a dominant at some sites. These sites currently dominated by Gardner saltbush show low cover of halogeton and comparatively low presence of remnant plants of halogeton. Sites with high abundance and vigorous growth of halogeton are those with major die-back of Gardner saltbush. These conditions strongly implicate halogeton as a controlling factor in die-off of Gardner

saltbush. The rather rapid conversion from Gardner saltbush to halogeton between the 1990s and mid 2000's indicates the drought of 2002 was a factor in the conversion. The impact of the 2002 drought is apparent by the low reading of 4.4 percent crown cover of Gardner saltbush in 2003 and the relatively high reading of 28.2 percent in 2004 at study site 6-16.

Halogeton is a poisonous plant that has caused major losses of sheep (Kingsbury 1964; Sharp and others 1990; Young 2002). It is essentially a non-forage plant, or at least it should not be included in evaluations of carrying capacity of rangelands of this area. Change induced by halogeton equates to reduced forage production. This marks a need to reduce stocking rates for livestock. Failure to do so will force livestock to use greater amounts of the remaining forage species.

Repeat photography can be highly effective in monitoring vegetation change in this setting. Line intercept measurements provide quantitative information, and in most cases 500 ft (152 m) of intercept was measured in less than 30 minutes. However, repeat photography alone demonstrates trend sufficiently well to leave little question that Gardner saltbush is being displaced by halogeton. In addition the photography demonstrates that magnitude of change is great enough that changes are needed in permitted livestock grazing.

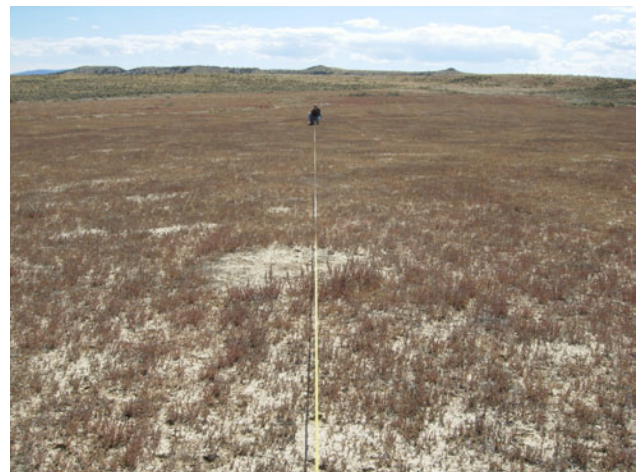


Figure 5. 10 Sept. 2009. Crown cover of Gardner saltbush was measured at 0%. Halogeton (reddish plant) has done more to eliminate Gardner saltbush in a decade than a century of livestock grazing.

Repeat photography has been demonstrated as a fast method to monitor trend in desert shrub communities (Sharp and others 1990) and in grass communities (Sharp 1992). This paper demonstrates that repeat photography, with notes, is adequate to determine trend and need for change in management in Gardner saltbush communities.

The photo record and measured trend in Gardner saltbush communities of this study indicate that monitoring intervals should be based on the rapidity of change in plant communities. In this case frequent monitoring should be considered. To better understand Gardner saltbush community dynamics, monitoring in years of high halogeton production and in years highly favorable for shrub growth seems important. The interval of 16 years (1993-2005) for line intercept measurements at study site 72-13 was too long to demonstrate how quickly the conversion took place. However, this interval was sufficient to document the extent of change. In plant communities of less frequent change, the interval of monitoring could be longer. It seems appropriate that mandates for frequency of monitoring should be based on frequency of change rather than on an arbitrary set interval.

Prairie dogs are sometimes considered agents of diversity. However, in the lower Green River Basin they foster the spread and establishment of halogeton as are other factors of disturbance. In this case they function as agents of lower diversity.

In an evaluation of cheatgrass and halogeton, Robocker (1961) made the following comment: "The concept of maintaining a status quo of climax, native perennial vegetation may now be forced into a re-evaluation by these exotic species." Trend in the area of this study supports the re-evaluation suggested nearly 50 years ago by Robocker (1961).

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Field Trip Overview: Habitat Loss and Plant Invasions in Northern Utah's Basin and Range

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ABSTRACT

An all-day field trip was conducted on May 19, 2010, as part of the 16th Wildland Shrub Symposium program. The tour consisted of Tour A and Tour B, which visited Utah's west desert shrublands in Park Valley (Tour A), and Bear River Mountains montane shrublands and Hardware Ranch Wildlife Management Area (Tour B). Both tours convened in the early afternoon at Golden Spike National Historic Site at Promontory, Utah, to visit salt desert-sagebrush revegetation research before the last stop to visit broom snakeweed/ sagebrush rangeland interaction research conducted on private lands adjacent to ATK facilities along Faust road in Box Elder County.

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Tour A – Utah's West Desert Shrublands, Park Valley

Ron Greer, Utah Division of Wildlife Resources (UDWR), hosted the first stop at a wildlife habitat restoration area just south of Highway 30 in Park Valley on the Overland Stage Route Road. A human ignited wildfire burned public and private land in the summer of 2005. Through the efforts of Utah's Watershed Restoration Initiative (UWRI), a partnership-driven effort to conserve, restore, and manage ecosystems in priority areas across the state, this area was successfully reseeded directly after the devastating wildfire. The seed mix included three varieties of crested wheatgrass (*Agropyron cristatum* [L.] Gaertn.), Siberian wheatgrass (*Agropyron fragile* [Roth] P. Candargy), Great Basin wildrye (*Leymus cinereus* [Scribn. & Merr.] A. Löve), Russian wildrye (*Psathyrostachys juncea* [Fisch.] Nevski), Snake River wheatgrass (*Elymus wawawaiensis* J. Carlson & Barkworth), three varieties of alfalfa (*Medicago sativa* L.), sainfoin (*Onobrychis viciifolia* Scop.), small burnet (*Sanguisorba minor* Scop.), fourwing saltbrush (*Atriplex canescens* [Pursh] Nutt.), and forage kochia (*Bassia prostrata* [L.] A.J. Scott.). The establishment of seeded species was highly successful and the results have enhanced this area of Park Valley for wildlife and biological diversity, water quality and yield for all uses, and provided opportunities for sustainable uses (figure 1).



Figure 1. Successful establishment of grasses by the Utah Division of Wildlife Resources and Utah Watershed Restoration Initiative in Park Valley, Utah following a 2005 wildfire.

Due to the timely actions of the UWRI regional team, consisting of Utah Partners for Conservation and Development members, conservation organizations, and local Park Valley stakeholders, who met to discuss priority conservation focus areas early on, this burned area was identified as a potential project where resources (funding, technical assistance, logistics support) could be implemented for restoring this area for sage grouse, deer, and livestock grazing habitat. It was critical to establish and enact an effort such as this to prevent the problems being faced just across fence boundaries and where the next stop took the field-trip participants.

Utah State University (Chris Call, Merilyn Hirsch, and Beth Fowers), USDA Agricultural Research Service (Tom Monaco and Justin Williams), and Private landowners (Royce Larsen and Ken Spackman) showcased their research demonstration areas located four miles south of Park Valley, Utah and Highway 30 in the second stop. Burned by the 2005 wildfire, this area was previously dominated by Wyoming big sagebrush and greasewood (*Sarcobatus vermiculatus* [Hook.] Torr.) with low species diversity in the shrub understory. This site was not reseeded after the fire and converted to a cheatgrass (*Bromus tectorum* L.) dominated landscape thereafter (figure 2).



Figure 2. Cheatgrass dominated areas where revegetation demonstration areas were established as part of the Area-Wide project in 2008.

Because it was dominated by cheatgrass, this area was chosen for demonstration research areas as part of the USDA Agricultural Research Service-funded Ecologically Based Invasive Plant Management (EBIPM) Area-Wide Project. Settlement of Park Valley began in the 1870s but really took off during the land boom of the 1910s. Settlers in Park Valley raised cattle and sheep. Livestock grazing was instrumental in the introduction and spread of invasive plant species in the Great Basin. Settlement was also accompanied by a great deal of land clearing to obtain homestead patents and for agriculture. The fallowed fields and cleared lands abandoned by homesteaders were staging areas of disturbed soil that harbored invasive species. Cheatgrass has greatly expanded in Park Valley since the early 1980s. In the last decade, wildfires in 1999 and 2005

promoted large expanses of rangelands dominated primarily by cheatgrass. Sandberg bluegrass (*Poa secunda* J. Presl) and squirreltail (*Elymus elymoides* [Raf.] Swezey) do occur, but generally at less than two percent ground cover. Attempts to seed crested wheatgrass and other perennials have been highly successful in certain areas if seeding occurs immediately after fires. The areas chosen for the EBIPM demonstration studies have been unsuccessfully seeded or never been seeded before. Evaluation of four treatments (intensive cattle grazing, prescribed fire, herbicide application, and drill seeding), alone and in combinations at large scales (10 to 30 acres) at this site were implemented to determine their effectiveness in modifying ecological processes and promoting a transition from a cheatgrass-dominated state to a perennial species dominated state.

Research led by Lesley R. Morris, USDA Agricultural Research Service, Logan, Utah, on historic dry farming impacts in Park Valley area was the focus of the third stop (figure 3). Across the arid West, dry farming (agriculture without irrigation) helped fuel a land rush of new homesteads after 1909. Homesteaders cleared sagebrush, plowed and harrowed the soils, and planted grains in hopes of making a living. Although successful in some areas, most of the dry farms failed and many people lost everything. The impacts of this historic land use can still be seen in aerial photos nearly 100 years after cultivation. Research objectives are to evaluate how site history (dry farming) has influenced rangeland vegetation and soil nearly a century after being cultivated. If this site history influences present conditions, it is likely to have similar influences on future management outcomes. Comparisons were made of vegetation and ground cover in historically dry-farmed areas to adjacent land outside of the historically cultivated fields at six paired sites across three ecological sites. Results of current research indicate that historic dry farming has had long-lasting impacts on vegetation and ground cover across different ecological sites that could influence key ecosystem properties. Understanding the legacies of this land use has important applications for invasive species management, ecological site classification, livestock producers and land management.



Figure 3. A typical shrubland site that was historically impacted by dry farming in the early 1900s. The recovery of native species in these areas has been variable across ecological sites.

Tour B – Bear River Mountains, Montane Shrublands, Hardware Ranch Wildlife Management Area

Dan Christensen and Darren Deblois of the UDWR lead the first two stops at the Hardware Ranch Wildlife Management Area (WMA) and mahogany forests located 15 miles up the Blacksmith Fork Canyon. This area has been a popular site for discussion of many issues such as wildlife politics, shrub management, and wildlife needs. Hardware Ranch WMA and surrounding land is an important range site for wildlife, especially in winter, when it is heavily utilized. High quality forage and habitat encourage wildlife to stay on wildlands and off farms and urban areas. The WMA visit showcased a mosaic of various species and size classes of shrubs in a background of forbs and grasses, each with a different utility for wildlife. The implications of changing plant communities (species, age classes, and densities), wildlife requirements, animal impacts on shrub communities, and shrub stand influence on the watershed were discussed. Of special interest and concern to this region is curl-leaf mountain mahogany (*Cercocarpus ledifolius* Nutt.), which is good forage for all classes of browsing animals in both summer and winter and is one of the few browse species that meets or exceeds the protein requirements for wintering big game animals (figure 4). It was noted that moose have a high preference for curl-leaf mountain mahogany stands in winter.

Currently curl-leaf mountain mahogany stands are mature with many populations in decline, yet lack regeneration of new stands. It appears that this species does germinate, but fails to establish well in wildland settings. One successful, but expensive tool used to establish shrubs in critical areas has been to plant containerized plants. Problems with containerized plants include water demands of plants with established leaf area, but with small roots, and the attractiveness of the plant material to foraging wildlife. Less expensive materials and methods of establishing shrubs are needed to improve wildlife habitat.



Figure 4. Discernible browse line on curl-leaf mountain mahogany tree.

Combined Afternoon Tour – Golden Spike National Historic Site And Broom Snakeweed / Sagebrush Rangeland Interactions

Eugene Schupp and Jan Summerhays of Utah State University lead a visit to their salt desert-sagebrush shrubland revegetation research south of the parking lot at the visitor center of Golden Spike National Historic Site (GSNHS) (figure 5). Golden Spike National Historic Site in Box Elder County, Utah, marks the location of the completion of the transcontinental railroad in 1869. Historically a sagebrush-steppe ecosystem, this site and its surrounding land have been subject to over a century of ground disturbing activities such as grazing and agriculture. These stressors have led to degradation of sagebrush ecosystems and the loss of understory perennial grasses and forbs. Current vegetation of

GSNHS consists of sagebrush grassland dominated by basin big sagebrush (*Artemisia tridentata* Nutt. ssp. *tridentata*), rubber rabbitbrush (*Ericameria nauseosa* [Pall. ex Pursh] G.L. Nesom & Baird), and purple three-awn (*Aristida purpurea* Nutt.). Disturbed areas along old railroad lines and roads have high concentrations of cheatgrass, common sunflower (*Helianthus annuus* L.), and broom snakeweed (*Gutierrezia sarothrae* [Pursh] Britton & Rusby).

One of the primary missions of the National Park Service is to conserve the natural and cultural resources and values of the national park system for the enjoyment of this and future generations. Restoring the existing vegetation community to resemble 1869 has limitations. Current research is

aimed at determining methods of reincorporating perennial grasses into the understory. However, due to the incidence of cultural resources and artifacts within Golden Spike National Historic Site, park management prohibits the use of ground-disturbing activities such as drill seeding. As such, all seeding must be done via aerial broadcasting, a non-disturbing method of seed distribution. A primary goal of the experiment is to search for ways to increase the success of aerial broadcast seeding. Restoration treatments at GSNHS were implemented with the specific purpose of manipulating soil nutrients and other resource conditions to favor perennial grass establishment while addressing some of the factors that contribute to cheatgrass dominance.



Figure 5. Eugene Schupp (at left) from Utah State University addressing tour participants at research plots located on the Golden Spike National Historic Site at Promontory, Utah.

The final stop of the afternoon was hosted by Michael Ralphs, USDA Agricultural Research Service (ret.), Logan, Utah and Chris Call of Utah State University. They illustrated broom snakeweed/sagebrush rangeland interactions along a well-defined fenceline contrast east of ATK facilities (Corinne, Utah) on Faust Valley Road (figure 6). A 5-year (2002-2006) study was initiated following grazing and fire disturbances on an Upland Gravelly Loam ecological site to evaluate broom snakeweed invasion in different plant communities. Broom snakeweed (*Gutierrezia sarothrae*) is an aggressive native invasive species that thrives after disturbance in semi-arid rangelands of the western U.S. The site originally had two plant communities: a 'sagebrush-bunchgrass' community that was grazed by cattle in alternate years in fall and winter, which was dominated by bluebunch wheatgrass and an open stand of Wyoming big sagebrush; and a 'sagebrush' community that was grazed in spring each year, which removed the bunch grasses, leaving a dense stand of Wyoming big sagebrush with an understory of Sandberg's bluegrass. Portions of these two plant communities were burned in a wildfire in 2001, removing the sagebrush, and creating two additional communities. By the end of their study, the burned portion of the sagebrush-bunchgrass community became a 'bluebunch wheatgrass' dominated community, and the burned portion of the sagebrush community became a 'snakeweed' dominated community. Mature snakeweed plants that existed in

the sagebrush-bunchgrass community died in 2003, due to competition from bunchgrasses during drought conditions. Snakeweed was eliminated in the bluebunch wheatgrass community by the wildfire in 2001, and did not reestablish. Snakeweed density and cover remained constant in the sagebrush community. Snakeweed cover increased from 2 to 31 percent in the snakeweed community, despite the presence of Sandberg bluegrass. The data were used to evaluate and update the current Upland Gravelly Loam (Wyoming big sagebrush) ecological site description and its state-and-transition model to reflect vegetation changes associated with snakeweed invasion.



Figure 6—Various vegetation states within close proximity with the Upland Gravelly Loam ecological site in northwestern Utah.

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