Special Report 1085
June 2008

Range Field Day 2008
Progress Report

Oregon State University | Agricultural Experiment Station

USDA United States Department of Agriculture
Agricultural Research Service

For additional copies of this publication

Eastern Oregon Agricultural Research Center (EOARC)
67826-A Hwy 205
Burns OR 97720
541-573-8900

Department of Rangeland Ecology and Management
Strand Agricultural Hall
Oregon State University
Corvallis OR 97331
541-737-3341

A PDF of the publication will be accessible from the EOARC website at
http://oregonstate.edu/dept/EOARC/

Agricultural Experiment Station
Oregon State University

Special Report 1085
June 2008

Range Field Day 2008 Progress Report

Eastern Oregon Agricultural Research Center
Department of Range Ecology and Management
Oregon State University
Agricultural Research Service, U.S. Department of Agriculture

This report is a cooperative effort of the Department of Rangeland Ecology and Management, Oregon State University, and the Eastern Oregon Agricultural Research Center. Any mention of commercial products does not constitute endorsement by Oregon State University or the USDA-Agricultural Research Service. The Eastern Oregon Agricultural Research Center, which includes Burns and Union Stations, is jointly operated and financed by the U.S. Department of Agriculture-Agricultural Research Service, Oregon State University, and the Oregon Agricultural Experiment Station.

Acknowledgments
Many private landowners, state, and federal agencies have provided funds and/or use of lands to perform the research presented in this report. We wish to thank the following private land owners for their contributions; Tim and Bill Lowery, Mike Stanford, Fred and Debbie Otley, Larry and Sue Otley, the Mullin Ranch, and the Lamb Ranch. Federal land management agencies to be recognized are the Burns, Lakeview, Alturas, Prineville, and Owyhee field offices of the USDI-Bureau of Land Management. Other federal support has been received from U.S. Department of Agriculture-Agricultural Research Service and the Joint Fire Science Program. State support has been provided by the Oregon Agricultural Experiment Station and Idaho Department of Lands. We wish to thank Georjanna Pokorney, Sally Olson-Edge, and Ariel Ginsburg for editing and preparation of this report.

TABLE OF CONTENTS

AUTHORS

JON BATES, Rangeland Scientist, Eastern Oregon Agricultural Research Center, USDA-Agricultural Research Service, Burns, Oregon

BRETT BINGHAM, Range Technician, Eastern Oregon Agricultural Research Center, USDA-Agricultural Research Service, Burns, Oregon

DAVID BOHNERT, Range Animal Nutritionist, Eastern Oregon Agricultural Research Center, Oregon State University, Burns, Oregon

CHAD BOYD, Rangeland Scientist, Eastern Oregon Agricultural Research Center, USDA-Agricultural Research Service, Burns, Oregon

JOHN BUCKHOUSE, Extension Watershed Specialist/Professor, Oregon State University Department of Rangeland Ecology and Management, Corvallis, Oregon

MICHAEL CARPINELLI, National Resource Conservation Service, Albuquerque, New Mexico

KIRK DAVIES, Rangeland Scientist, Eastern Oregon Agricultural Research Center, USDA-Agricultural Research Service, Burns, Oregon

TIM DEBOODT, County Extension Agent/Associate Professor, Oregon State University Extension Service, Crook County, Prineville, Oregon

PAUL DOESCHER, Professor, Oregon State University, Department of Forest Science, Corvallis, Oregon

MICHEAL FISHER, Associate Professor, Forestry Department, Central Oregon Community College, Bend, Oregon

DAVE GANSKOPP, Rangeland Scientist, Eastern Oregon Agricultural Research Center, USDA-Agricultural Research Service, Burns, Oregon

STUART HARDEGREE, Plant Physiologist, Northwest Watershed Research Center, USDA-Agricultural Research Service, Boise, Idaho

DUSTIN JOHNSON, Range Extension, Harney County, Oregon State University, Burns, Oregon

JANE MANGOLD, Rangeland Scientist, Eastern Oregon Agricultural Research Center, USDA-Agricultural Research Service, Burns, Oregon

RICK MILLER, Professor Rangeland Ecology, Eastern Oregon Agricultural Research Center, Oregon State University, Burns, Oregon

FRED PIERSON, Research Leader, Northwest Watershed Research Center, USDA-Agricultural Research Service, Boise, Idaho

JAIME RATCHFORD, Research Assistant, Oregon State University Department of Rangeland Ecology and Management Resources, Corvallis, Oregon

KIMBERLY REEVER-MORGHAN, Eugene, Oregon

ED RHODES, Research Associate, Center for Natural Resource Information Technology, Department of Ecosystem Science and Management, Texas A&M University, College Station

BREANNA SABIN, Graduate Student, Oregon State University, Department of Rangeland Ecology and Management, Corvallis, Oregon

ROB SHARP, Rangeland Specialist, Bureau of Land Management, Burns District, Oregon

ROGER SHELLY, Weed Ecologist, Eastern Oregon Agricultural Research Center, USDA-Agricultural Research Service, Burns, Oregon

JOHN SWANSON, Rangeland Specialist, Bureau of Land Management, Prineville District, Prineville, Oregon

TONY SVEJCAR, Research Leader and Rangeland Scientist, Eastern Oregon Agricultural Research Center, USDA-Agricultural Research Service, Burns, Oregon

ANDREW TIERNEY, Graduate Student, Oregon State University, Department of Rangeland Ecology and Management, Corvallis, Oregon

EDWARD VASQUEZ, Post-doctoral Research Assistant, USDA-Agricultural Research Service, Burns, Oregon

Forward

The 2008 Range Field Day will focus on several subjects important to managing the Intermountain regions rangelands productively and ecologically. The main topics presented are livestock grazing management, medusahead ecology and management, and western juniper ecology and management.

Research on the management of livestock grazing in western rangelands continues to advance practices that properly utilize forage resources while maintaining or improving ecosystem values. This report addresses several grazing issues: fire and grazing interactions, resilience of riparian communities to defoliation, and developing a better understanding of factors influencing grazing patterns and seasonal movement of cattle in large pastures.

Medusahead, along with cheatgrass, are the most pressing management issues concerning the health of sagebrush grasslands. Integrative management that combines herbicide application, fire, and the reseeding of competitive plant species appears to be the most viable approach to revegetate medusahead infested rangelands, albeit an expensive treatment. Aggressive and comprehensive prevention programs that include early detection and eradication of medusahead remain the best means for conserving rangelands that are currently medusahead-free. This report highlights the impacts of several medusahead control applications as well as explaining why it is such a difficult and challenging weed to manage in Intermountain rangelands.

Western juniper remains a significant issue in the management of northern Great Basin rangelands. Much progress has been made in explaining the expansion of western juniper woodlands the past century, quantifying the impacts of woodland expansion on the ecosystem, and developing management tools to control juniper and restore Great Basin plant communities. Included in the report are evaluations on the magnitude of juniper expansion at a regional level, watershed impacts of woodlands, and several new approaches to juniper control using partial cutting and fire combinations to restore sagebrush grassland and aspen woodland.

If you have further questions on the topics of today’s field day we encourage you to contact the authors for further information. Finally, most of the projects presented at the Field Day and/or in the progress report have resulted from our interactions with private and public land-owners. The goals of our programs are to develop answers and strategies that maintain or enhance the many resources and services provided by our rangelands. Your input has and will continue to be a key component for developing research programs that are pertinent to your interests and concerns.

Grazing Management

Influence of Long-term Livestock Grazing Exclusion on the Response of Sagebrush Steppe Plant Communities to Fire

Kirk W. Davies, Tony J. Svejcar, and Jon D. Bates

SUMMARY

Livestock grazing of sagebrush steppe plant communities has been considered to have negative impacts because these communities did not evolve with large herbivores. The best management of these ecosystems has been assumed to be accomplished by mimicking historic conditions, such as application of prescribed fire. However, the introduction of invasive plants or altered environmental conditions has potentially changed the response of plant communities to fire disturbance. This study evaluated the effects of fire to plant community dynamics between grazed pasture and long-term grazing exclosures. Treatments were ungrazed (grazing excluded since 1936), ungrazed burned, grazed unburned, and grazed burned. Vegetation cover, density, and biomass production were measured in the 12th, 13th, and 14th year post-burning. Long-term grazing exclusion followed by burning resulted in a substantial cheatgrass invasion. The ungrazed burned treatment had the least perennial vegetation and greatest annual vegetation. The grazed burned treatment had the greatest amount of perennial herbaceous productivity and density among all treatments. Our results suggest that long-term grazing exclusion weakens the ability of Wyoming big sagebrush plant communities to tolerate fire and thus allows cheatgrass invasion. Low to moderate grazing by domestic livestock appears to be better management than grazing exclusion for maintaining sagebrush steppe.

INTRODUCTION & OBJECTIVES

Historic disturbance regimes are often reconstructed to direct ecosystem management. It has been assumed that the best management and restoration of ecosystems would be accomplished by mimicking historic disturbance regimes. However, some environmental conditions and biotic potentials have been irreversibly altered and, therefore, could potentially change the response of the plant community to disturbance. For example, climate change or invasive plants could result in a different response to disturbance than expected. Thus, the outcome of reintroducing historic disturbances in native plant communities is not well understood, especially with the threat of invasive plants.

To evaluate reintroducing historic disturbances, we investigated the implications of reintroducing fire with and without long-term livestock grazing in Wyoming big sagebrush plant communities in the northern Great Basin. Wyoming big sagebrush plant communities have been estimated to have a historic fire return interval of 50-100 years and evolved with few large herbivores resulting in little grazing pressure.

The impacts of livestock grazing on plant communities that did not evolve with large numbers of herbivores are generally considered negative. These plant communities are expected to be intolerant of livestock grazing pressure. However, light to moderate utilization by domestic livestock has been demonstrated to have minimal impacts on sagebrush plant communities.

Understanding the impacts of different disturbances patterns on Wyoming big sagebrush plant communities is important because most of these communities are grazed by domestic livestock, are at risk of burning, and provide valuable habitat for wildlife. Ungrazed plant communities are probably more likely to burn because of a buildup of fine fuels. If Wyoming big sagebrush communities are going to be managed according to their historic disturbance regime, some late seral sagebrush plant communities would be prescribe burned and domestic livestock would be removed. The impact of returning the historic disturbance regime to Wyoming big sagebrush steppe remains uncertain with the current threat of cheatgrass invasion.

METHODS

The study was conducted at the Northern Great Basin Experimental Range (NGBER) in southeastern Oregon about 35 miles west of Burns, Oregon, USA. Treatments were: 1) ungrazed unburned, 2) ungrazed burned, 3) grazed unburned, and 4) grazed burned. Ungrazed treatments were established in 1936 with the erection of 5 acre grazing exclosures. Data collected in 1937 revealed no differences in the densities of Sandberg bluegrass, large perennial grasses, annual grasses, perennial forbs, and annual forbs between inside and outside the exclosures. Cheatgrass was not present inside or outside the exclosures in 1937. Areas adjacent to the exclosures were grazed by cattle until 1990. Grazing pressure was low to moderate. In late September of 1993, prescribed burning treatments were applied as strip-head fires using drip torches. Vegetation cover, density, and biomass production were sampled in 2005, 2006, and 2007, the 12^{th}, 13^{th}, and 14^{th} year post-burning, respectively.

RESULTS

Cover

The interaction between burning and grazing treatments influenced cover of all herbaceous functional groups (Fig.1). Large perennial bunchgrass cover was greatest in the grazed burned treatment and lowest in the ungrazed burned treatment. Cheatgrass cover was 8.6 times greater in the ungrazed burned treatment than any of the other treatments. Similarly, annual forb cover was greatest in the ungrazed burned treatment, while perennial forb cover was lowest in this treatment. Moss cover was lowest in the ungrazed burned treatment and highest in the ungrazed unburned treatment.

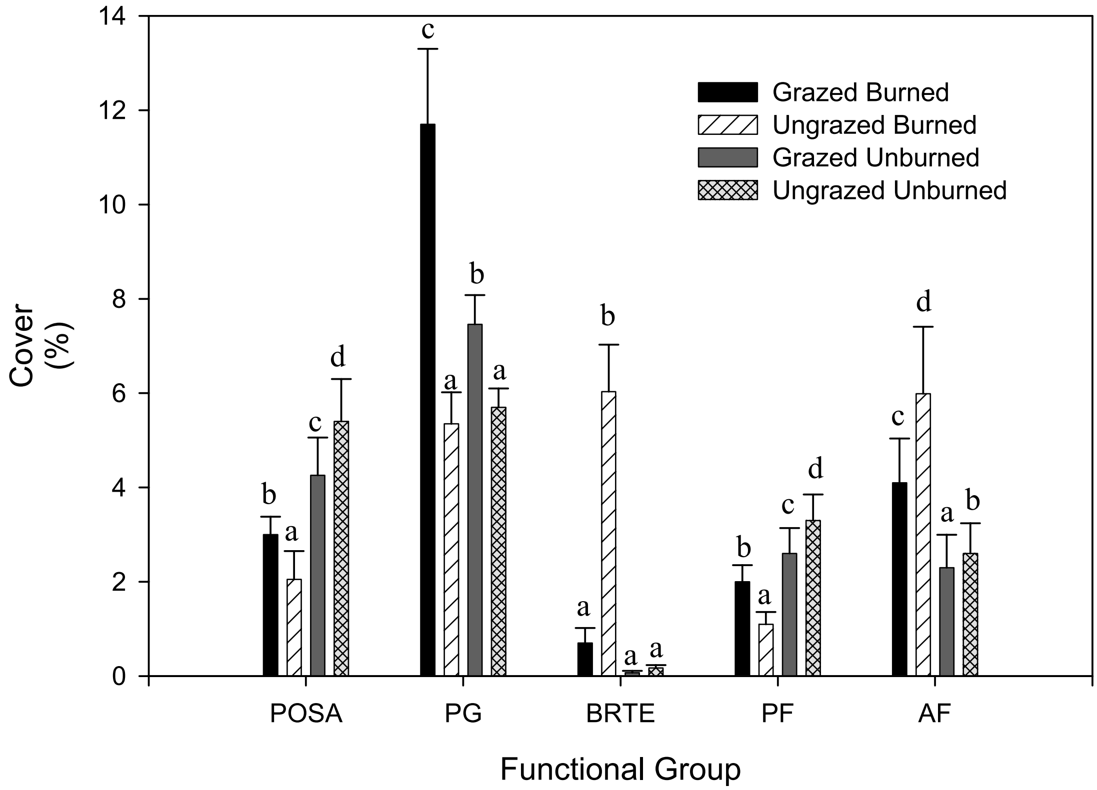


Figure 1. Functional group cover (mean + S.E.) of the treatments averaged over 2005, 2006, and 2007 at the Northern Great Basin Experimental Range. POSA = Sandberg bluegrass, PG = tall perennial bunchgrass, BRTE = cheatgrass, PF = perennial forb, and AF = annual forb. Ungrazed = livestock excluded since 1937, Grazed = livestock were allowed to graze until 1990, Burned = prescribed fall burning in 1993, and Unburned = no prescribed burning. Different lowercase letters indicated a significant difference ($P < 0.05$) among treatments.

Density

Large perennial bunchgrass density was lowest in the ungrazed burned treatment and highest in the grazed burned treatment with an approximately 1.9-fold difference between the two treatments (Fig. 2). Burning decreased perennial bunchgrass density in the ungrazed treatment, but did not influence bunchgrass density in the grazed treatment. Burning increased cheatgrass density in the ungrazed treatment. Perennial forb density was decreased by burning, but was not influenced by grazing. Burning generally increased green rabbitbrush density, however, the increase in density was largest in the ungrazed treatment.

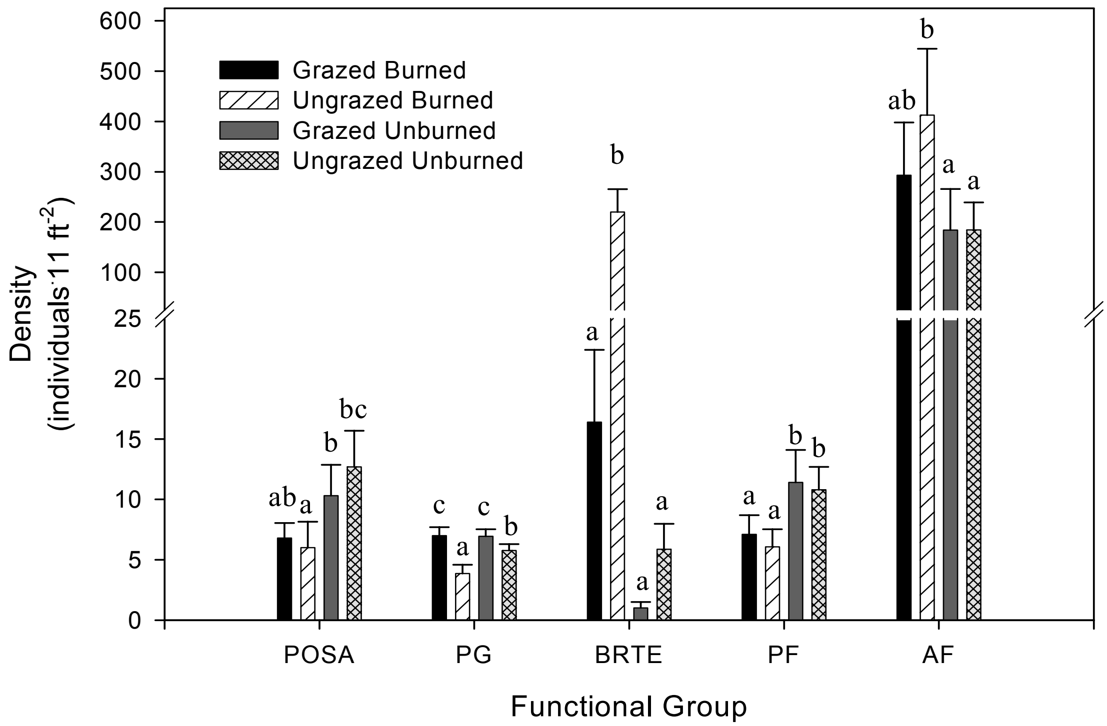


Figure 2. Functional group density (mean + S.E.) of the treatments averaged over 2005, 2006, and 2007 at the Northern Great Basin Experimental Range. POSA = Sandberg bluegrass, PG = tall perennial bunchgrass, BRTE = Cheatgrass, PF = perennial forb, and AF = annual forb. Ungrazed = livestock excluded since 1937, Grazed = livestock were allowed to graze until 1990, Burned = prescribed fall burning in 1993, and Unburned = no prescribed burning. Different lowercase letters indicated a significant difference (P < 0.05) among treatments.

Biomass

Large perennial bunchgrass production generally increased with burning (Fig. 3). Bunchgrass production increased more with burning in the grazed compared to the ungrazed treatment. Burning the grazed treatment increased perennial bunchgrass production 1.6-fold. Cheatgrass biomass production increased more than 49-fold after burning the ungrazed treatment. Perennial forb biomass production decreased 3-fold when the ungrazed treatment was burned. Biomass production of annual forbs increased with burning. Annual forb production was lowest in the ungrazed unburned treatment and highest in the ungrazed burned treatment. In the ungrazed burned treatment, cheatgrass produced more biomass than all the perennial herbaceous vegetation combined. Combining cheatgrass and annual forb production reveals that annuals produced 2.8-fold more biomass than perennial herbaceous vegetation in the ungrazed burned treatment. The ungrazed burned treatment was the only treatment to produce more annual than perennial herbaceous vegetation biomass.

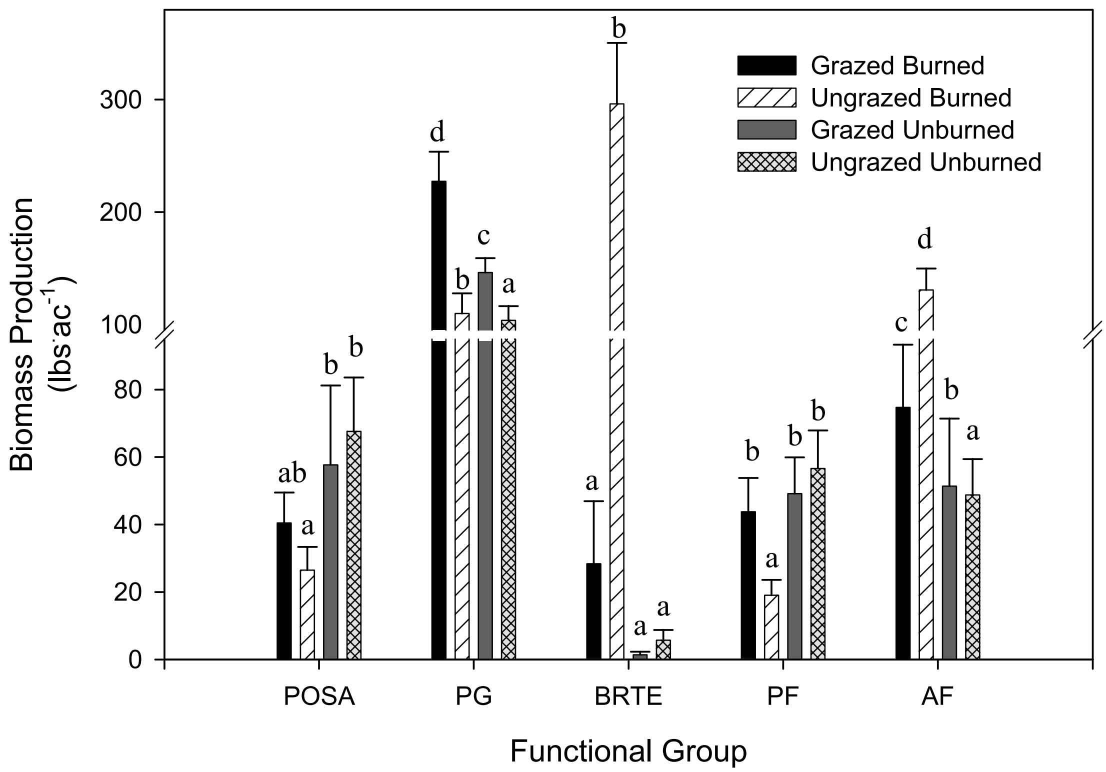


Figure 3. Functional group biomass production (mean + S.E.) of the treatments averaged over 2005, 2006, and 2007 at the Northern Great Basin Experimental Range. POSA = Sandberg bluegrass, PG = tall perennial bunchgrass, BRTE = Cheatgrass, PF = perennial forb, and AF = annual forb. Ungrazed = livestock excluded since 1937, Grazed = livestock were allowed to graze until 1990, Burned = prescribed fall burning in 1993, and Unburned = no prescribed burning. Different lowercase letters indicated a significant difference (P < 0.05) among treatments.

MANAGEMENT IMPLICATIONS

The lack of livestock grazing in the big sagebrush plant communities weakened the ability of the perennial herbaceous vegetation to tolerate fire. This could be the result of accumulation of fuels or a loss of mechanisms important to tolerating disturbances. Low to moderate livestock grazing appears to be beneficial to the long-term sustainability of Wyoming big sagebrush plant communities. Preventing grazing to protect sagebrush plant communities and sagebrush obligate wildlife species may actually result in their loss. The large increase in cheatgrass with grazing protection followed by fire is a concern. On a larger area, the increase in cheatgrass would greatly increase the likelihood of higher fire frequency due to more fine fuel loads and continuity. The increased fire frequency would be detrimental to native vegetation and sagebrush obligate wildlife species.

REFERENCES

Baker, W.L. 1994. Restoration of landscape structure altered by fire suppression. Conservation Biology 8:763-769.

Belsky, A.J., and D.M. Blumenthal. 1997. Effects of livestock grazing on stand dynamics and soils in upland forests of the Interior West. Conservation Biology 11:315-327.

Courtois, D.R., B.L. Perryman, and H.S. Hussein. 2004. Vegetation change after 65 years of grazing and grazing exclusion. Journal of Range Management 57:574-582.

Davies, K.W., J.D. Bates, and R.F. Miller. 2007. Short-term effects of burning Wyoming big sagebrush steppe in southeast Oregon. Rangeland Ecology and Management 60:515-522.

Eckert, R.E., and J.S. Spencer. 1986. Vegetation response on allotments grazed under rest-rotation management. Journal of Range Management 39:166-174.

Eckert, R.E., and J.S. Spencer. 1987. Growth and reproduction of grasses heavily grazed under rest-rotation management. Journal of Range Management 40:156-159.

Fleischner, T.L. 1994. Ecological costs of livestock grazing in western North America. Conservation Biology 8:629-644.

Mack, R.N., and J.N. Thompson. 1982. Evolution in steppe with few large, hooved mammals. The American Naturalist 119:757-773.

Manier, D.J., and N.T. Hobbs. 2006. Large herbivores influence the composition and diversity of shrub-steppe communities in the Rocky Mountains, USA. Oecologia 146:641-651.

Rice, B., and M. Westoby. 1978. Vegetative responses of some Great Basin shrub communities protected against jackrabbits or domestic stock. Journal of Range Management 31:28-34.

Rickard, W.H. 1985. Experimental cattle grazing in a relatively undisturbed shrubsteppe community. Northwest Science 59:66-72.

West, N.E., F.D. Provenza, P.S. Johnson, and M.K. Owens. 1984. Vegetation change after 13 years of livestock grazing exclusion on sagebrush semidesert in west central Utah. Journal of Range Management 37:262-264.

Wright, H.A., and A.W. Bailey. 1982. Fire Ecology: United States and Southern Canada. John Wiley & Sons Inc. New York, NY, USA. pp. 159-160.

Grazing After Fire in the Sagebrush-Steppe

Jon Bates, Ed Rhodes, Kirk Davies, and Rob Sharp

SUMMARY

This study evaluated cattle grazing impacts over four growing seasons after prescribed fire on Wyoming big sagebrush steppe in eastern Oregon. Treatments included no grazing on burned and unburned sagebrush steppe, two summer grazing applications after fire, and two spring grazing applications after fire. Treatment plots were burned in fall 2002. Grazing was applied in 2003-2005. Vegetation responses to treatments were evaluated by quantifying plant cover, density, standing crop, production, and measuring perennial grass seed production. Standing crop and seed production were greater in the ungrazed burn treatment than all grazed burn treatments; however, these differences did not affect community recovery after fire. Herbaceous response variables (cover, density, and production), bare ground, and litter cover did not differ among grazed and ungrazed burn treatments. Burn treatments (grazed and ungrazed) had greater herbaceous cover, standing crop, herbaceous production, and seed production than the unburned treatment by the second or third year after fire. The results demonstrated that properly applied livestock grazing after low severity fire will not slow or reduce the recovery of plant communities in big sagebrush steppe.

INTRODUCTION

In sagebrush rangelands of the western United States, fire has been a natural and prescribed disturbance temporarily shifting vegetation from shrub-grass co-dominance to grass dominance. There is limited information on the impacts of grazing to community dynamics following fire in the sagebrush ecosystem. In 2001, we developed a study to evaluate post-fire herbaceous recovery of sagebrush steppe in eastern Oregon as influenced by time of grazing reintroduction. Four moderate grazing treatments after fire were compared to burned and unburned treatments in Wyoming big sagebrush steppe. Grazing treatments were comprised of two summer and two spring scenarios. We predicted that (1) grazing in the spring starting the second year after fire (one year of rest) would reduce herbaceous plant recovery compared to other grazed and no-graze burn treatments; (2) summer grazing treatments and spring grazing (with two years of post-fire rest) would have similar herbaceous recovery levels compared to the no-graze burn treatment; and 3) herbaceous response in the summer grazing treatments, spring grazing (with two years' post-fire rest) treatment, and the no-graze burn treatment would exceed the unburned treatment within three years following fire.

METHODS

The study was conducted at the Northern Great Basin Experimental Range, 35 miles west of Burns, Oregon, USA. Elevation at the site is 4,500 ft and slope less than 2%. Annual precipitation has averaged 11.9 inches since the 1930's. Wyoming big sagebrush was the dominant shrub and green rabbitbrush was a secondary shrub. The understory was co-dominated by Idaho fescue and Thurber's needlegrass. Sandberg's bluegrass, bluebunch wheatgrass, prairie Junegrass, and bottlebrush squirreltail were present as subdominant grasses. Prescribed burning was applied in late September and early October, 2002. Fires were complete across burn plots, killing nearly 99% of the Wyoming sagebrush present.

Cattle grazing impacts to post-fire recovery of herbaceous vegetation was evaluated over four growing seasons. Thirty 4.5-5.0 acre plots were established in 2001. There were six treatments applied and all treatments were replicated 5 times. The treatments were;

SUMMER 1: graze the first 2 years after fire in early August 2003 and 2004.
SUMMER 2: graze the second and third summer after fire in August 2004 and 2005.
SPRING 1: graze the second and third spring after fire in May 2004 and 2005.
SPRING 2: graze the third spring after fire in May 2005. This treatment is equivalent to many post-fire grazing programs in the region.
BURN: no grazing after fire.
UNBURNED: no fire and no grazing.

The summer grazing treatments took place in early August when herbaceous plants were largely dormant or had completed that year's growth cycle. The spring grazing treatments took place in early to mid-May during vegetative and early boot stages of growth of the main bunchgrass species (Idaho fescue, and Thurber's needlegrass). No grazing was applied in 2006 as this was the main response year we used to compare herbaceous recovery among the treatments. Treatment plots were individually fenced to control livestock. Grazing was managed to remove 40-50% of herbaceous standing crop in all grazed treatments. This is considered a moderate to slightly higher than moderate level of use in the sagebrush steppe. Vegetation responses to treatments were evaluated by quantifying plant cover, density, clipping for standing crop and production; and perennial grass seed production.

RESULTS

Fire Severity; Fire initially caused a reduction in cover of herbaceous perennials and green rabbitbrush. However, densities of herbaceous perennials and green rabbitbrush were unaffected by the fire indicating a fire of low severity. Sagebrush was severely affected by fire as most individuals were killed. Fire intensity was sufficient to reduce sagebrush to stumps less than 4 inches in height.

Utilization; Utilization in both summer grazing treatments and the SPRING 1 treatment in 2004 were close to the targeted level of 50%. In spring 2005, herbage was growing rapidly and grazed plants were re-grew quickly while livestock were still grazing in the treatment plots. Measured utilization showed only light use (25%) in SPRING 1 and SPRING 2 treatments in 2005.

Bare Ground; Bare ground increased the first year after fire in all treatments that were burned (Fig. 1). Bare ground was greater after fire in burn treatments (grazed and ungrazed) than the UNBURNED in 2003. By the second year after fire (2004) there were no longer any differences among treatments and in the burned treatments levels of bare ground did not differ from pre-burn conditions.

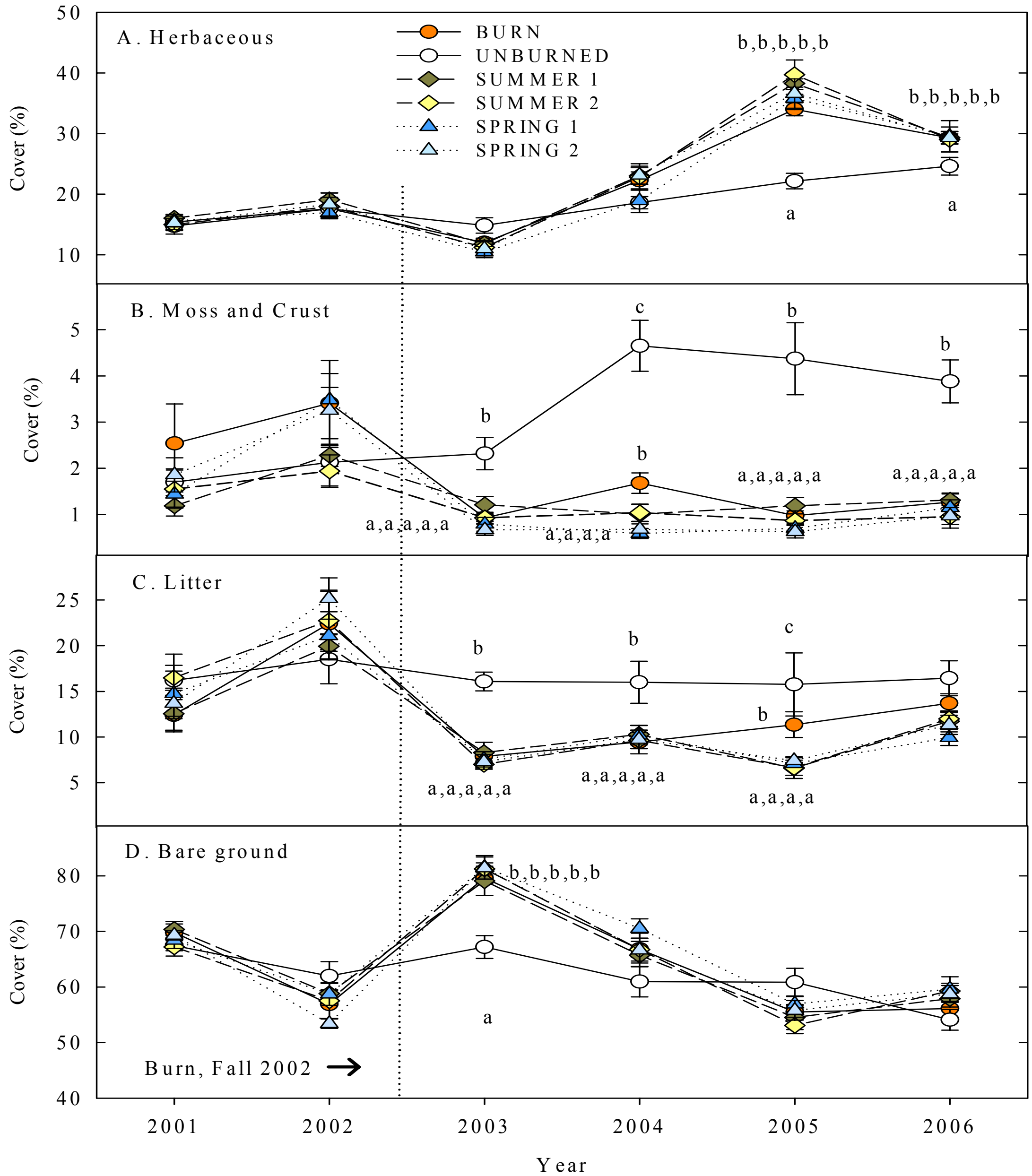


Figure 1. Ground cover values for the various burn and grazing treatments in Wyoming big sagebrush steppe, 2001-2006; A) herbaceous, B) litter, C) moss and biotic crust, and D) bare ground and rock.. Values represent means $\pm$ one standard error. Different lower-case letters indicate significant differences ($p<0.05$) among the treatments within year.

Litter; Litter was reduced by burning and litter cover was greater in the UNBURNED treatment compared to most of the burned grazed treatments the first three years after fire. (Fig. 1). By 2006 treatments did not differ in litter cover.

Moss and Biotic Crust; Moss and other biotic crust increased in the UNBURNED treatment between 2002 and 2003 and remained 3.5-4 times greater than the burned treatments (grazed and ungrazed) (Fig. 1). Moss comprised most of the cover in this group and was mainly found within grass clumps and under sagebrush. At this point fire, not grazing, negatively impacted moss and biotic crust.

Herbaceous Cover; Prior to burning, cover values did not differ among treatments. The first year after burning (2003) cover was greater in the UNBURNED than all burned treatments (grazed and ungrazed) for most response variables (Fig. 1). By the third growing season (2005) after fire herbaceous cover was twice as great in the burned treatments (grazed and ungrazed) than the UNBURNED. Cover remained greater in all burned treatments than the UNBURNED treatment in 2006, although the magnitude of difference was only about 20%. In all treatments, perennial grass cover was greater than pre-burn levels in 2005 and 2006 but did not differ among treatments. Annual forb cover was greater in burned (grazed and ungrazed) treatments from the second through fourth year after fire (2004-2006). Cheatgrass cover remained low (<0.01%) throughout the study on all treatments.

Perennial Plant Density and Species Presence; Perennial plant densities remained largely unaffected by treatment. Fire caused no measurable mortality to perennial bunchgrasses. There were no differences among treatments in the numbers of species. Numbers of herbaceous species increased in all treatments in 2005 and 2006 when precipitation was greater than average.

Herbaceous Standing Crop; Herbaceous and functional group standing crop did not differ among treatments prior to burning (Figs. 2 and 3). By the third (2005) and fourth (2006) growing season after fire herbaceous and perennial grass standing crop was greater in all the burned treatments (grazed and no-graze) than the UNBURNED treatment. Standing crop in the BURN treatment was about twice as great as the UNBURNED treatment and was greater than all the burned-grazed treatments (except the SUMMER 2 treatment) in 2005 and 2006. The lower amount of standing crop in the burn-grazed treatments than the BURN treatment in 2005 and 2006 reflects the removal of herbage by livestock that reduces the amount of carry over biomass from previous year's growth. Sandberg's bluegrass standing crop was reduced in all burn treatments after fire. In 2005, Sandberg's bluegrass standing crop was lowest in the SPRING 1 and SPRING 2 treatments, respectively, than other treatments. Perennial forb standing crop/production was reduced in all burn (grazed and ungrazed) treatments the first year after fire. Annual forb standing crop/production increased in all burn treatments (grazed and ungrazed) the second-growing season after fire and was greater than the UNBURNED treatment. Over 90% of annual forb production was comprised of pale alyssum, an introduced Old World weed.

Herbaceous Productivity; Production values provided a different perspective when evaluating impacts of grazing after fire because residual material from previous year's

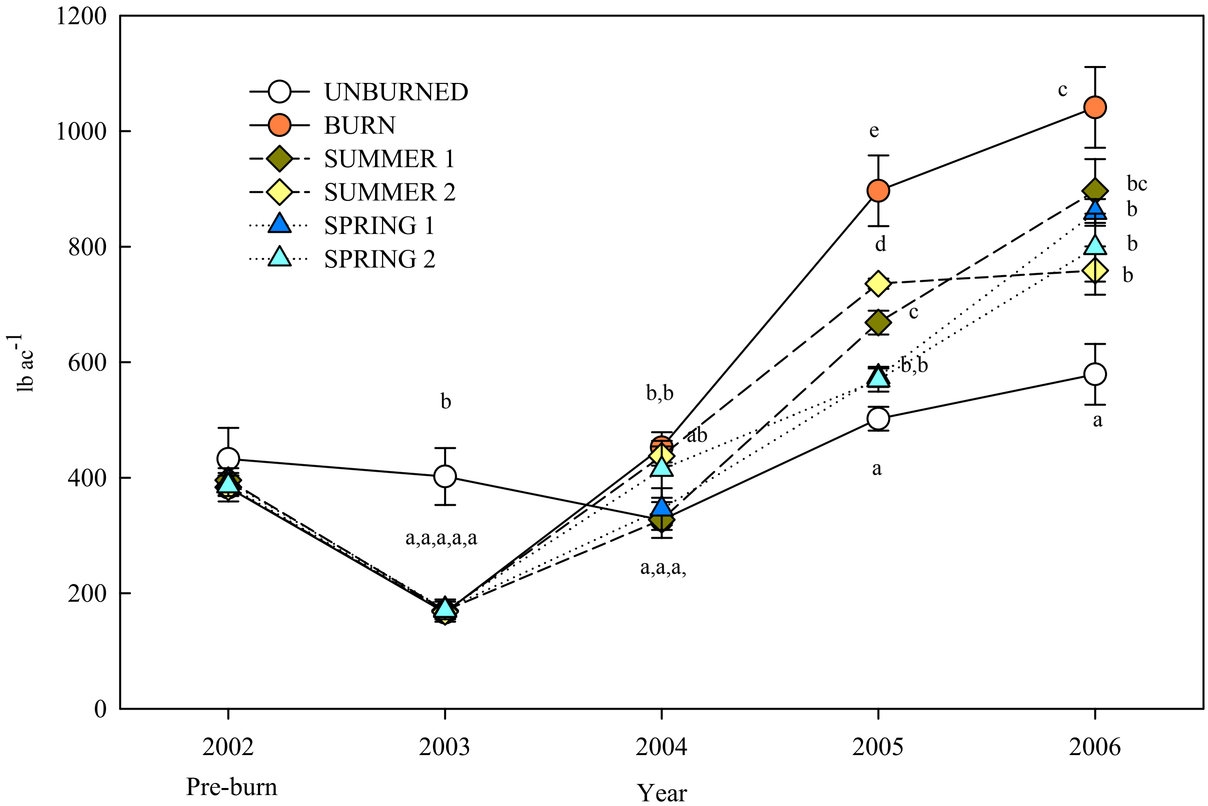


Figure 2. Herbaceous standing crop (lb ac^{-1}) for the various burn and grazing treatments in Wyoming big sagebrush steppe, 2001-2006. Values represent means $\pm$ one standard error. Different letters indicate significant differences among the treatments within year.

growth is removed prior to weighing (Fig. 4). In contrast to standing crop results, differences among the burn-grazed treatments and the BURN treatment were less apparent for total herbaceous and perennial grass production in 2005 and 2006. Neither summer grazing treatments differed from the BURN treatment for these response variables. Production in the SPRING 1 and SPRING 2 treatments was less than the BURN and both summer treatments because of herbage removal by livestock in 2005. Despite biomass removal, herbaceous and perennial grass production was greater in SPRING 1 and SPRING 2 treatments than the UNBURNED treatment in 2005. In 2006, herbaceous and perennial grass production did not differ among the burn (grazed and ungrazed) treatments. Herbaceous and perennial grass production was about two times greater in the BURN and burned-grazed treatments than the UNBURNED treatment in 2006.

Seed Production; Total seed production was greater in 2005 than 2004 for all treatments. Seed production was greater in the BURN and all burn-grazed treatments than the UNBURNED treatment in 2005 (Fig. 5). Within the burn treatments grazing influenced total seed production as well seed production of individual species. The BURN treatment had greater total seed production than the SUMMER 1, SPRING 1, and SPRING 2 treatments.

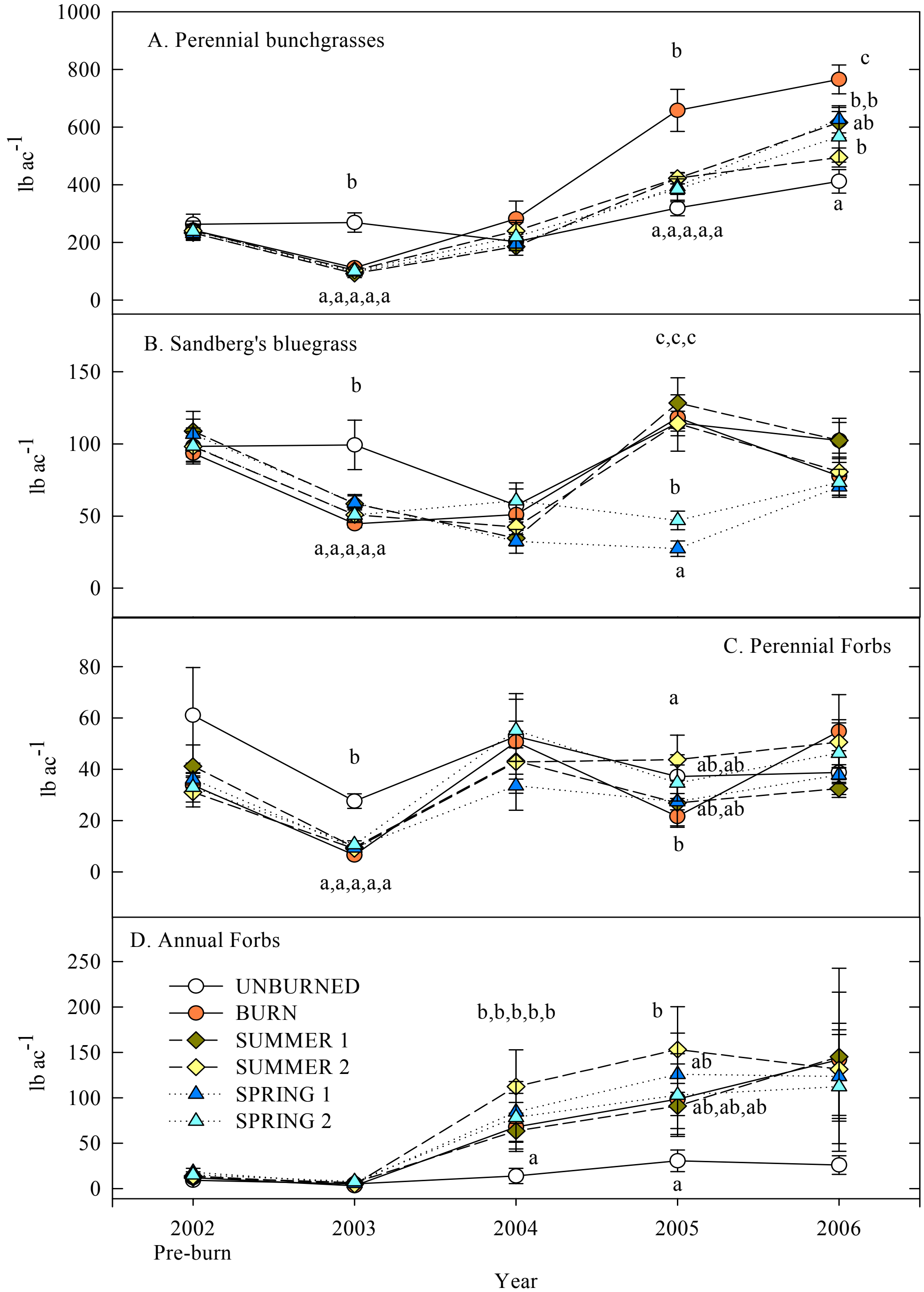


Figure 3. Functional group standing crop values (lb ac^{-1}) for the various burn and grazing treatments in Wyoming big sagebrush steppe, 2001-2006; A) Perennial bunch grasses; B) Sandberg's bluegrass; C) Perennial Forbs; and D) Annual forbs. Values represent means $\pm$ one standard error. Different lower-case letters indicate significant differences among the treatments within year.

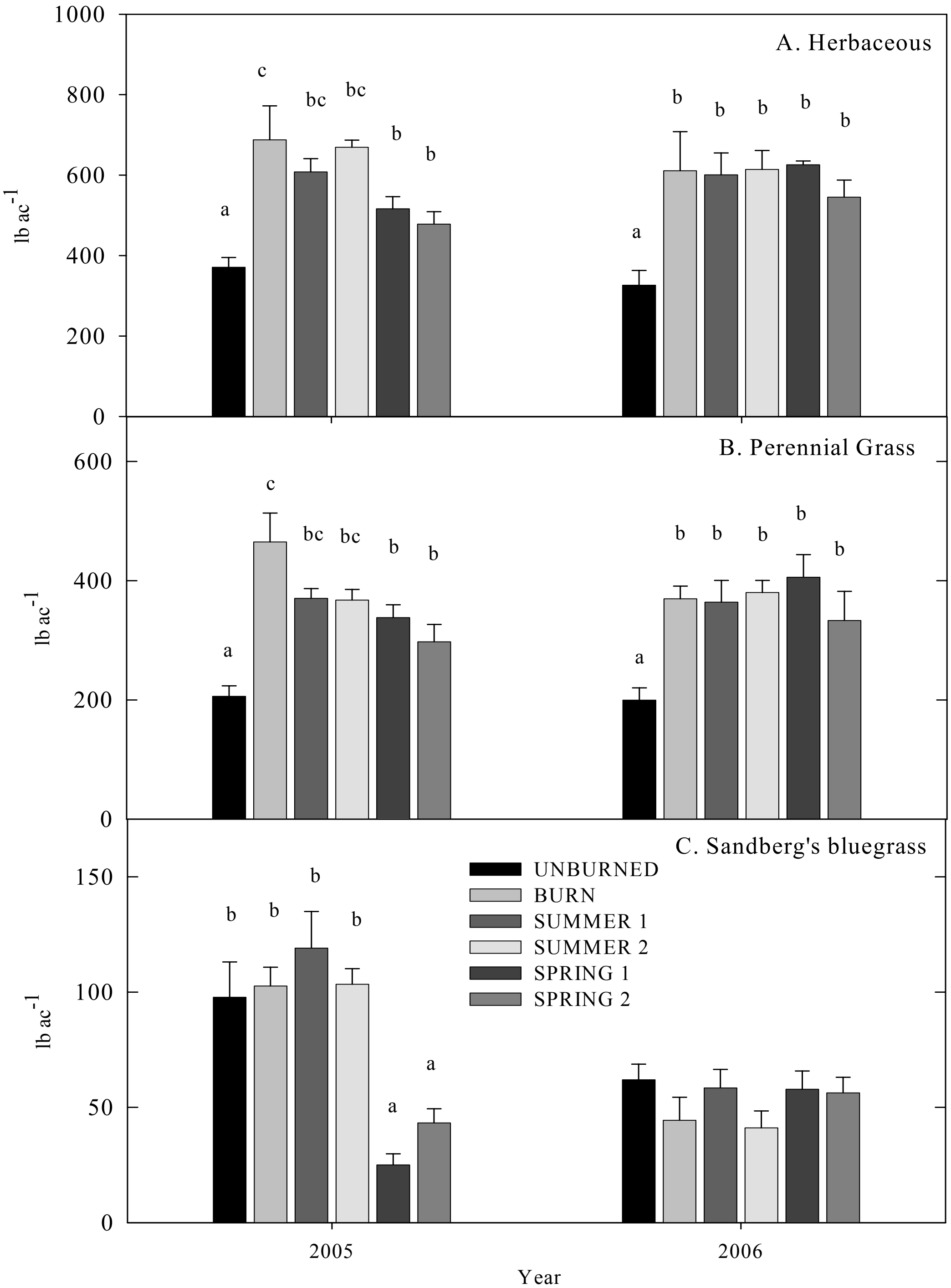


Figure 4. Production values (lb ac^{-1}) for A) Herbaceous, B) Perennial bunchgrasses, and C) Sandberg's bluegrass for the various burn and grazing treatments in Wyoming big sagebrush steppe, 2005-2006; Values represent means ± one standard error. Different lower-case letters indicate significant differences ($p<0.05$) among the treatments within year.

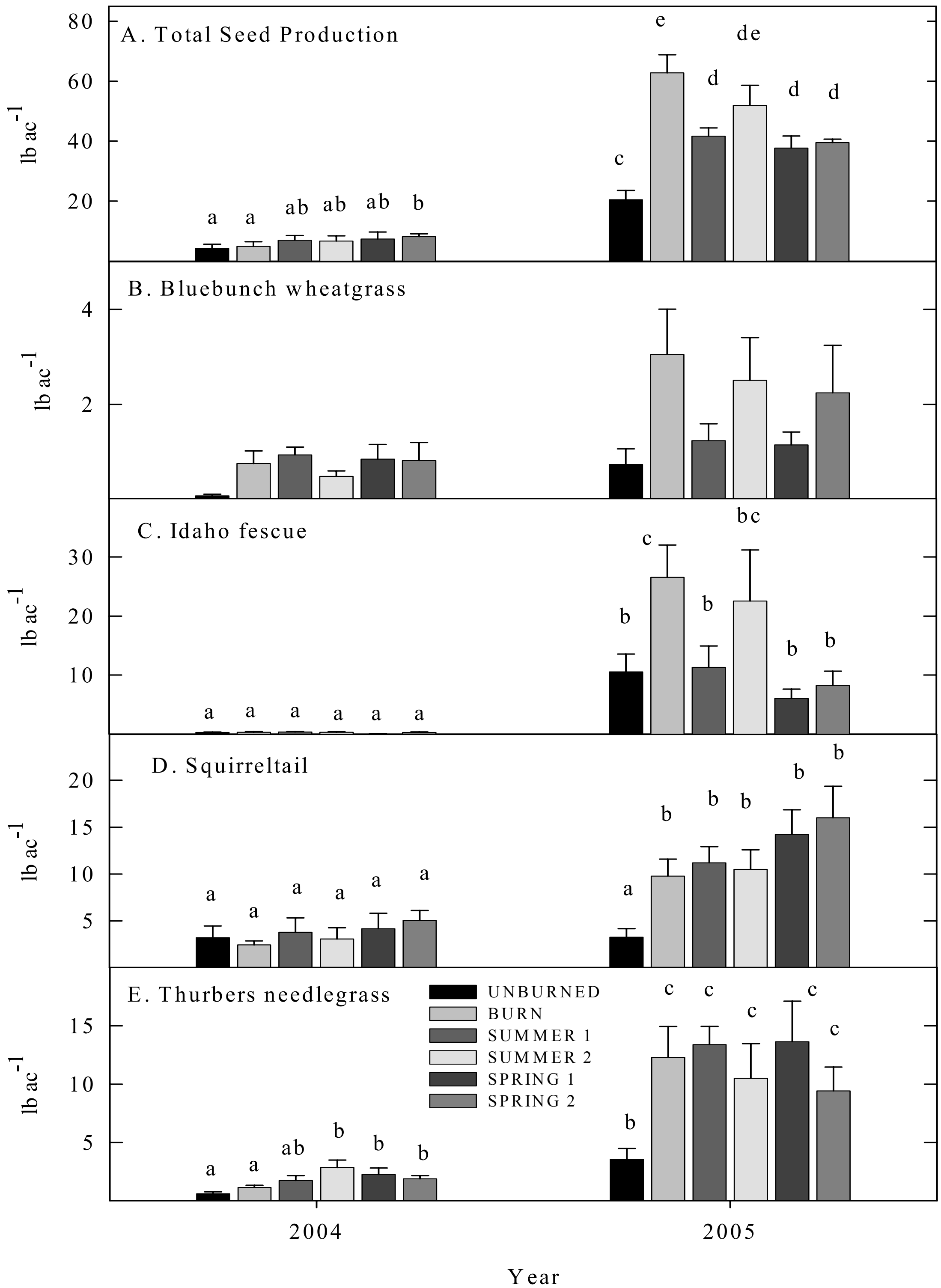


Figure 5. Seed production values (lb ac^{-1}) for the various burn and grazing treatments in Wyoming big sagebrush steppe, 2005-2006, Northern Great basin Experimental Range; A) Total Perennial Bunchgrasses; B) Bluebunch wheatgrass; C) Idaho fescue; D) Squirreltail; and E) Thurber's needlegrass.

DISCUSSSION and CONCLUSIONS

The results from this study indicate that moderate grazing following completion of the first growth cycle after low severity fire does not limit herbaceous recovery in big sagebrush steppe. Treatment differences were considered minor when evaluating community recovery after fire. For most measured variables (bare ground, litter cover, and herbaceous cover, density, and production) there were no differences among grazed and ungrazed burn treatments, particularly in the response year of 2006.

The response of herbaceous vegetation after fire, whether grazed or not, was comparable to results from other post-fire work in the sagebrush system. Herbaceous cover, standing crop and production in the BURN and burn-grazed treatments equaled or exceeded the UNBURNED treatment by the second or third year after fire. A factor for the rapid and progressive herbaceous response was the low severity of the prescribed fire. Fire can negatively impact bunchgrass species by killing individuals and reducing plant size especially species with densely packed culms such as Idaho fescue and Thurber's needlegrass. These were the most common perennial grass species on our study sites; however, neither species was reduced in density, and both species demonstrated a positive response by the second year after fire.

Cheatgrass and other exotic species are a major threat to maintaining Wyoming big sagebrush communities. Our results demonstrated that burning, with or without grazing, can successfully stimulate herbaceous native species and not result in an increase of cheatgrass. There were likely two reasons for the lack of a cheatgrass response; the pre-fire community was largely composed of native perennials and there was limited mortality of perennial grasses. High mortality of perennials would like have created conditions that would allow cheatgrass to increase. The main exotic that increased and comprised the bulk of forb productivity and composition after fire was pale alyssum. Information on the competitive abilities of alyssum is not available, although it likely interferes with native annual forbs since root characteristics and phenology appear to be similar. The presence of high densities of alyssum does not appear to obstruct the recovery of perennial grasses.

Moss and biological crust had not recovered to pre-burn levels by the fourth year after fire. Recovery of mosses and biological crusts after fire varies depending on species and plant community composition but recovery appears to be a lengthy process. In our study fire rather than grazing appears to have had the main impact on moss and biotic crust.

MANAGEMENT IMPLICATIONS

The primary goals of post-fire ecosystem management are the recovery of ecological processes (hydrologic function, energy and resource capture), preferred plant communities, wildlife habitat, and economic use. In sagebrush steppe plant communities these goals are achieved by recovering the system to one comprised of perennial grasses, forbs, and shrubs. This study demonstrated that properly applied livestock grazing after one growth cycle following fire will not slow or reduce the recovery of herbaceous plant communities in

Wyoming big sagebrush steppe. The study also demonstrated that grazing rest the first 2 years after fire to encourage herbaceous recovery may not be necessary in all situations.

The results and interpretations of this study must be considered under the conditions which it was conducted. The trials were performed on a single site; fire caused minimal mortality to perennial bunchgrasses; there was a lack of a significant weed presence; and grazing protocols were strictly controlled. One or more of these elements will vary in other situations and generate a host of different post-fire recovery scenarios. Study plots were small and we managed to obtain uniform grazing use. However, livestock tend not to graze uniformly in large pastures in the Great Basin, as distance to water and topography results in areas of high, moderate, low, and non-use. Grazing after fire in larger pastures and for longer duration would likely result in areas of differential use and rates of herbaceous recovery.

The summer grazing treatments provided the most robust outcome regarding herbaceous recovery as our results were in agreement with recent post-fire grazing and defoliation trials. Moderate grazing use after perennial grass dormancy the first couple summers after fire should not to reduce the recovery of post-fire herbaceous communities in sagebrush steppe.

This is the first such study of spring grazing in sagebrush steppe after fire and trials only evaluated defoliation during vegetative and early boot stages of growth of the larger perennial bunchgrasses. As alluded to earlier, defoliation of grasses in later boot or flower stages might have resulted in slower herbaceous recovery. At this point, grazing sagebrush steppe in the spring the first two years after fire should be applied cautiously.

ACKNOWLEDGMENTS

We would like to thank numerous summer crew members for assistance in the field and laboratory. Lori Ziegenhagen did much of the data compilation and running the field crews in 2003 and 2004. Many thanks to Matt Carlon, Lynn Carlon, and Clare Poulsen for helping on the burn team.

REFERENCES

BLM. 2007. Emergency fire rehabilitation handbook. United States Bureau of Land Management, Washington, DC.

Bruce, L.B., B. Perryman, K. Conley, and K. McAdoo. 2007. Case Study: Grazing management on seeded and unseeded post fire public rangelands. The Professional Animal Scientist 23:285-290.

Bunting, S.C., R. Robberecht and G.E. DeFosse. 1998. Length and timing of grazing on post burn productivity of two bunchgrasses in an Idaho experimental range. International Journal of Wildland Fire 8:15-20.

Jirik, S.J. and S.C. Bunting. 1994. Post-fire defoliation response of *Agropyron spicatum* and *Sitanion hystrix*. International Journal of Wildland Fire 4:77-82.

Defoliation Impacts on Above and Below-Ground Production in a Riparian Sedge Community

Chad S. Boyd and Tony J. Svejcar

SUMMARY

In spite of the interest in grazing impacts on riparian systems, there is limited information on root response of riparian sedges to grazing. We evaluated both above-ground and below-ground productivity in plots clipped in either June or July to a 4-inch stubble height. The study was designed as a randomized block with four replications and was conducted during 2004 and 2005. Root growth cores were used to harvest annual root growth, and plots were clipped to estimate above-ground end-of-season standing crop. Water tables were higher in 2005 for most of the growing season. Clipping treatments reduced end-of-season above-ground standing crop, but season-long production (clipped mass + end-of-season mass) was less impacted by clipping treatment. Root mass was only reduced by the July 2005 clipping treatment, and root length density was not significantly impacted by any treatment. In our study, root abundance was very high and resilient to clipping treatments.

INTRODUCTION

Previous research has provided guidance for above-ground regrowth of riparian sedge communities following variable timing and intensity of defoliation. This information serves as background material that can be used in developing management strategies to meet end-of-season stubble height objectives and provides guidance for maximizing above-ground production in grazed riparian systems. However, from a functional standpoint, the importance of below-ground production in riparian communities is at least equal to that of above-ground production. Roots are key to nutrient acquisition and soil structure, and in riparian systems function to hold banks in place during high flow events. Bank maintenance is particularly important because banks help maintain channel structure and the associated high water table necessary for riparian plant species. At present there is only scant information on the effects of defoliation on below-ground production in riparian sedge communities. The objective of this study was to determine the effects of clipping and clipping date on above- and below-ground production in a sedge-dominated riparian community.

METHODS

Data were collected during the growing seasons of 2004 and 2005. We used a randomized block design with four sites (i.e. blocks) adjacent to Nicoll Creek, Harney Co., Oregon. Sites were dominated by Nebraska sedge and were not grazed during the study. We used one clipping height (4-inches) and two clipping dates (June or July). Within a site we located eight, 40 x 20 inch plots. Adjacent plots were randomly assigned to experimental (clipped) or control (unclipped) treatments. Experimental plots were clipped in June or July. Six of the eight plots at one site and four of the eight plots at another site were grazed by cattle in early August of 2004 and were omitted from analyses for that year. We clipped all plots to ground level in October. All clipped vegetation was oven-dried and weighted.

Two 3-inch diameter cores were excavated to 11.8 inches deep in each plot and filled with sand in the fall of 2003 and 2004. Cores were harvested in September of 2004 and 2005 by driving a length of 2-inch chamfered PVC pipe into the center of the core. A shop-vac with an in-line collection reservoir was then used to evacuate the sand and root material from the PVC pipe (Fig. 1). Harvested roots and soil were separated in a root washer, scanned for root length density, oven dried, and weighed. Water table elevation was measured in PVC wells at bi-weekly intervals from May through September. Values were averaged across sites, within collection date and year.

Figure 1. Overview of root harvest technique. Ingrowth cores (3 inch-diameter) were excavated to a 11.8 inch-depth in streamside-plots during the fall and filled with sand. Cores were harvested after one year using a shop vacuum modified with a PVC suction attachment and collection reservoir.

RESULTS

Groundwater was at maximum depth in early to mid August (approximately 3 to 4 inches below the ground surface, Fig. 2). Groundwater levels were within the range of tolerance for wet sedge communities (i.e., water availability was not limiting factor sedge growth).

End-of-season above-ground standing crop decreased with clipping (average decline of 47% across treatments) for all time periods; particularly with July clipping (Fig. 3A). Season-long production (clipped mass + end-of-season standing crop) was less impacted by clipping and decreased (average decline of 15%) for clipped plots in July 2004 and June 2005 (Fig. 3B). Timing of clipping had no clear pattern of effect across years. Greater above-ground production in 2005 was associated with increased groundwater during the June-July period (Fig. 2). Root production and root length density were not affected by

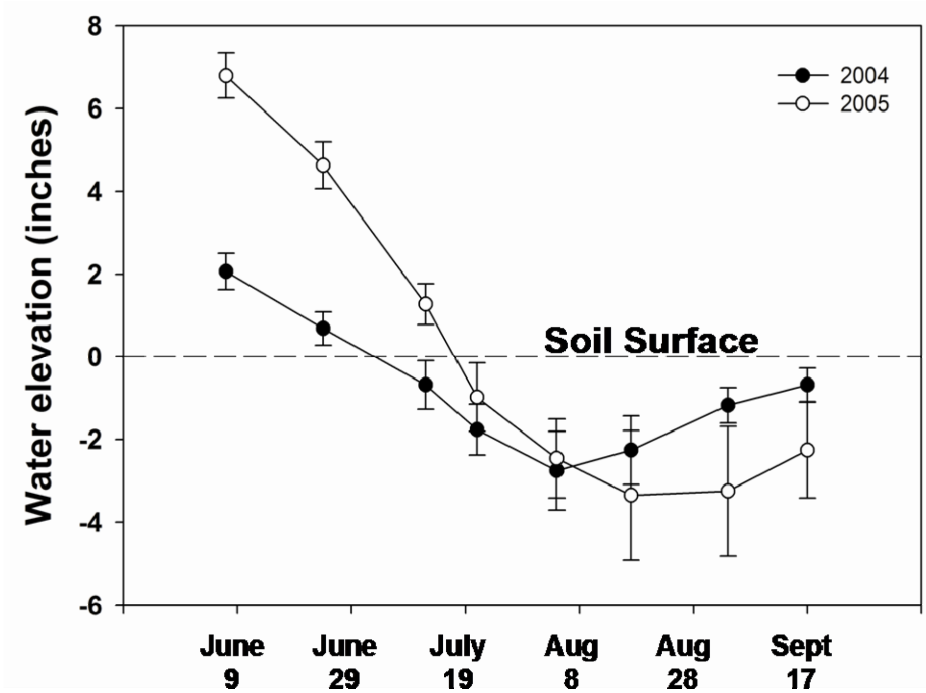


Figure 2. Water elevation for plots used in sedge clipping study. Elevations represent either the depth of standing water (positive values) or depth to ground water (negative values). Data were averaged over the 4 study sites, by date and within year.

clipping treatment with the exception of July, 2005 when root production in July-clipped plots decreased by 34% compared to unclipped plots (Fig. 4). Decreased root production in 2005 was associated with increased groundwater during the June-July period (Fig. 2).

MANAGEMENT IMPLICATIONS

An elevated water table was associated with increased above-ground and decreased below-ground production. Clipping to 4 inches can decrease end-of-season above-ground standing crop; however, season-long production is less impacted with no clear pattern of effect for timing of clipping. Root production was not strongly impacted by clipping treatment and appears to be resilient to clipping. Our study confirmed that root production in riparian sedge communities is very high compared to other ecosystems. Overall, our data suggest that below-ground production would not be impaired with moderate levels of herbivory in June or July.

REFERENCES

Boyd, C.S. and T.J. Svejcar. 2004. Regrowth and production of herbaceous riparian vegetation following defoliation. Journal of Range Management 57:448-454.

Clary, W.P. and W.C. Leininger. 2000. Stubble height as a tool for management of riparian areas. Journal of Range Management 46:493-497.

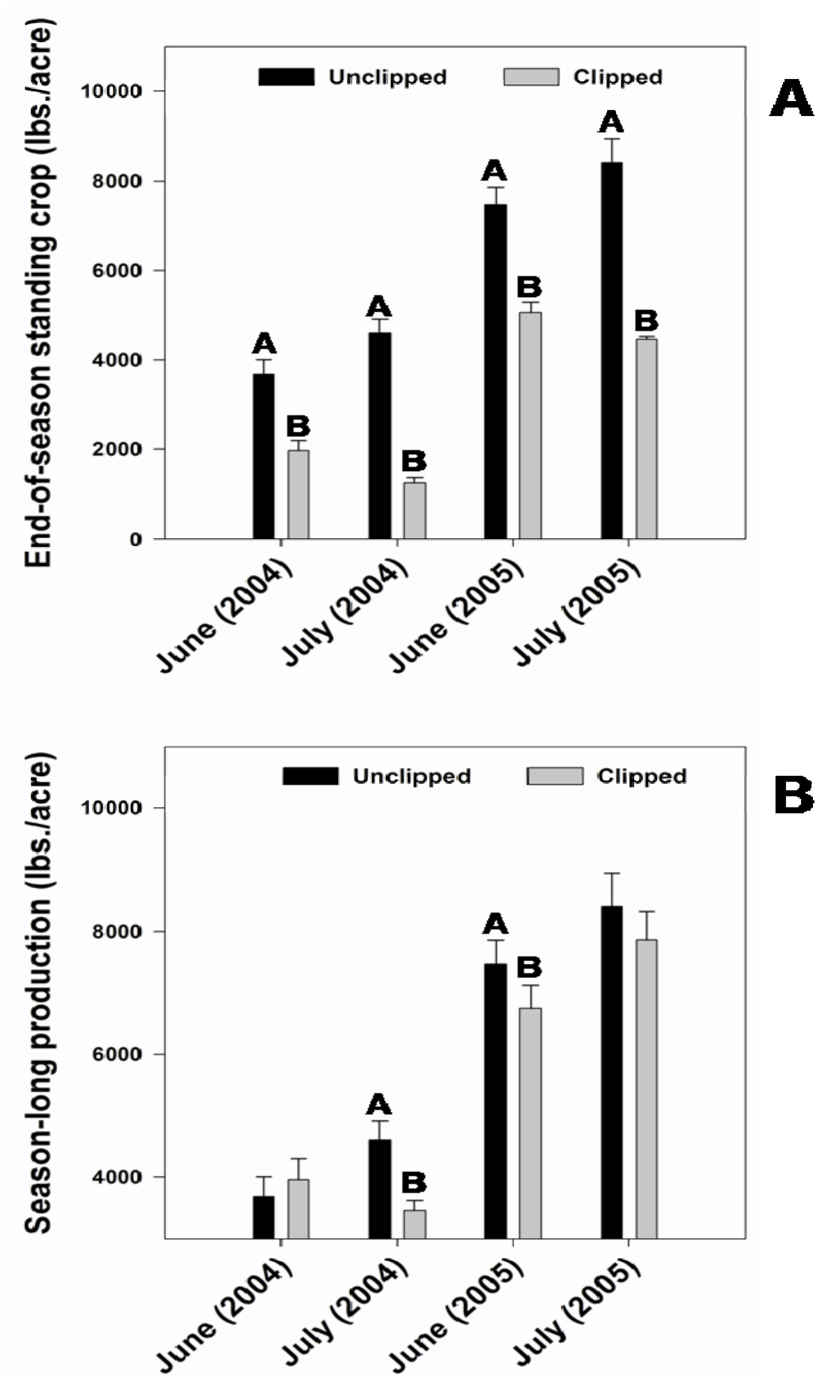


Figure 3. Effects of clipping (4-inch stubble height) and timing of clipping on above-ground production as measured by end-of-season standing crop (A) and season-long production (B) for sedge-dominated riparian plots at Nicoll Creek, Harney Co., Oregon. Differing letters within a bar pair indicate strong evidence of statistical difference.

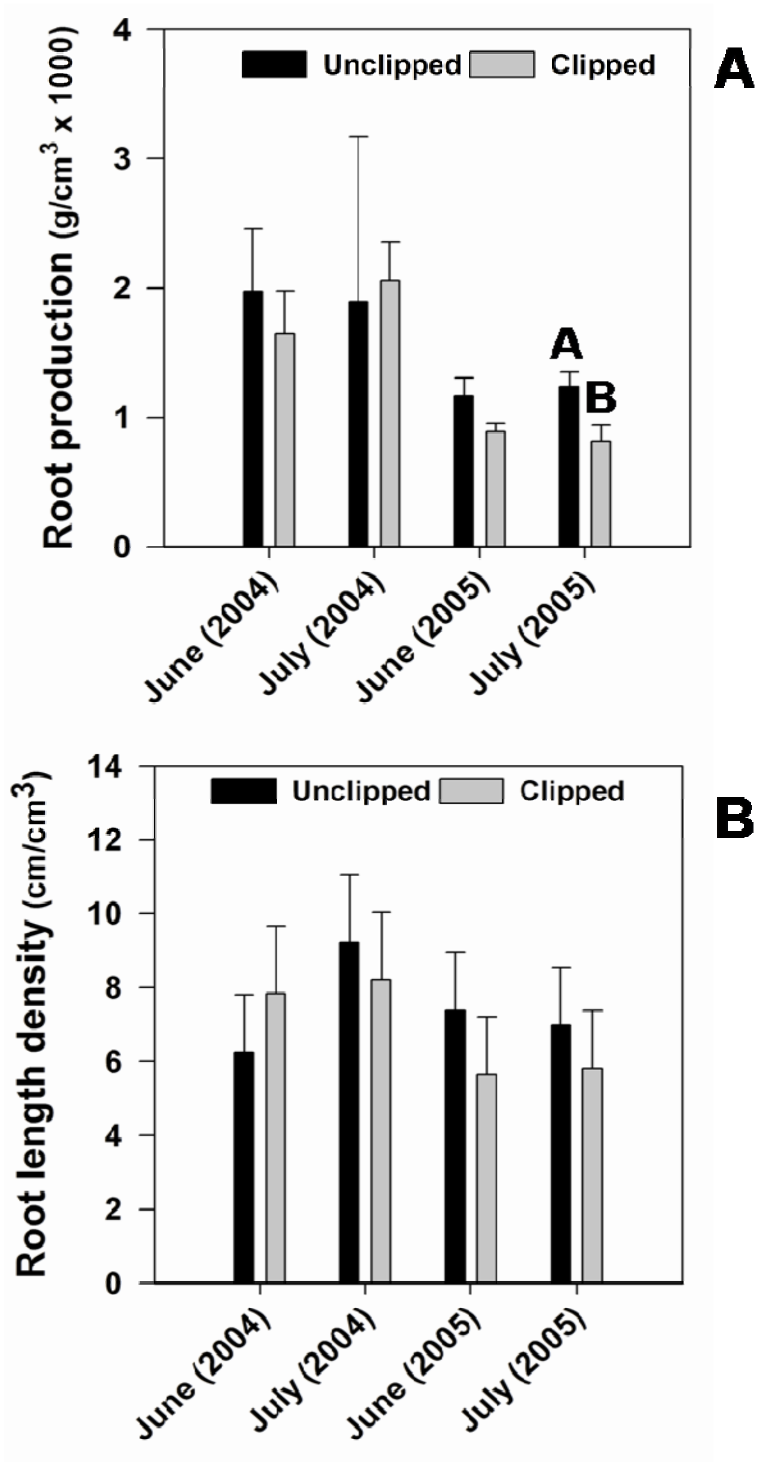


Figure 4. Effects of clipping (4-inch stubble height) and timing of clipping on root production as measured by weight (A) and root-length density (B) for sedge-dominated riparian plots at Nicoll Creek, Harney Co., Oregon. Differing letters within a bar pair indicate strong evidence of statistical difference.

Do Grazing Cattle Seek Nutritionally Superior Portions of Pastures?

Dave Ganskopp and Dave Bohnert

SUMMARY

This study evaluated the hypothesis that grazing cattle will most often frequent nutritionally superior portions of large pastures. Forage quantity/quality characteristics were mapped among three pastures and cattle grazing patterns subsequently tracked with GPS collars. Cattle preferred locations with higher than average crude protein and digestibility and areas with lower than average standing crop and neutral detergent fiber. These preferences likely explain adherence of cattle to historic grazing patterns and seasonal spatial moves of cattle in large pastures. Management practices that can affect desirable changes in forage quality, like mowing, burning, or prescribed grazing, can likely be used to attract stock to historically unused locales.

INTRODUCTION

On rangelands, uneven or unmanaged livestock distribution can adversely affect plant community composition, riparian function, or displace wildlife. These issues have historic precedents and are still a challenge for those managing rangelands today. A thorough understanding of factors affecting livestock distribution can help land and livestock managers avoid or moderate deleterious effects.

Optimum foraging theory suggests animals minimize energy expended and maximize energy returns when grazing, but foraging decisions by large animals at relevant temporal and spatial scales have not been studied. The objective of this research was to test the hypothesis that grazing cattle will seek nutritionally superior portions of pastures.

METHODS

The research was conducted in three 2000+ acre pastures on the Northern Great Basin Experimental Range 30 miles west of Burns, Oregon. Plant communities include a sparse western juniper overstory and a shrub layer dominated by Wyoming big sagebrush, mountain big sagebrush, or low sagebrush. Prominent grasses include bluebunch wheatgrass, Idaho fescue, or Sandberg's bluegrass depending on locale.

Four hundred fifty three coordinates were established in an offset grid spanning the 3 pastures (Figure 1). With this arrangement, each coordinate was equidistant (278 yds) from its six closest neighbors. Coordinates were downloaded to geographic positioning systems (GPS units), and personnel dispersed to each locale in mid June 2004. At each location, a frame was dropped and all standing grasses and forbs rooted therein were clipped to a 1-inch stubble. Subsequent forage quality assays for each sample included; crude protein (CP), neutral detergent fiber (NDF), and digestibility as indexed by *in situ* dry matter disappearance (ISDMD). Immediately after clipping, 15 cow/calf pairs were released to each

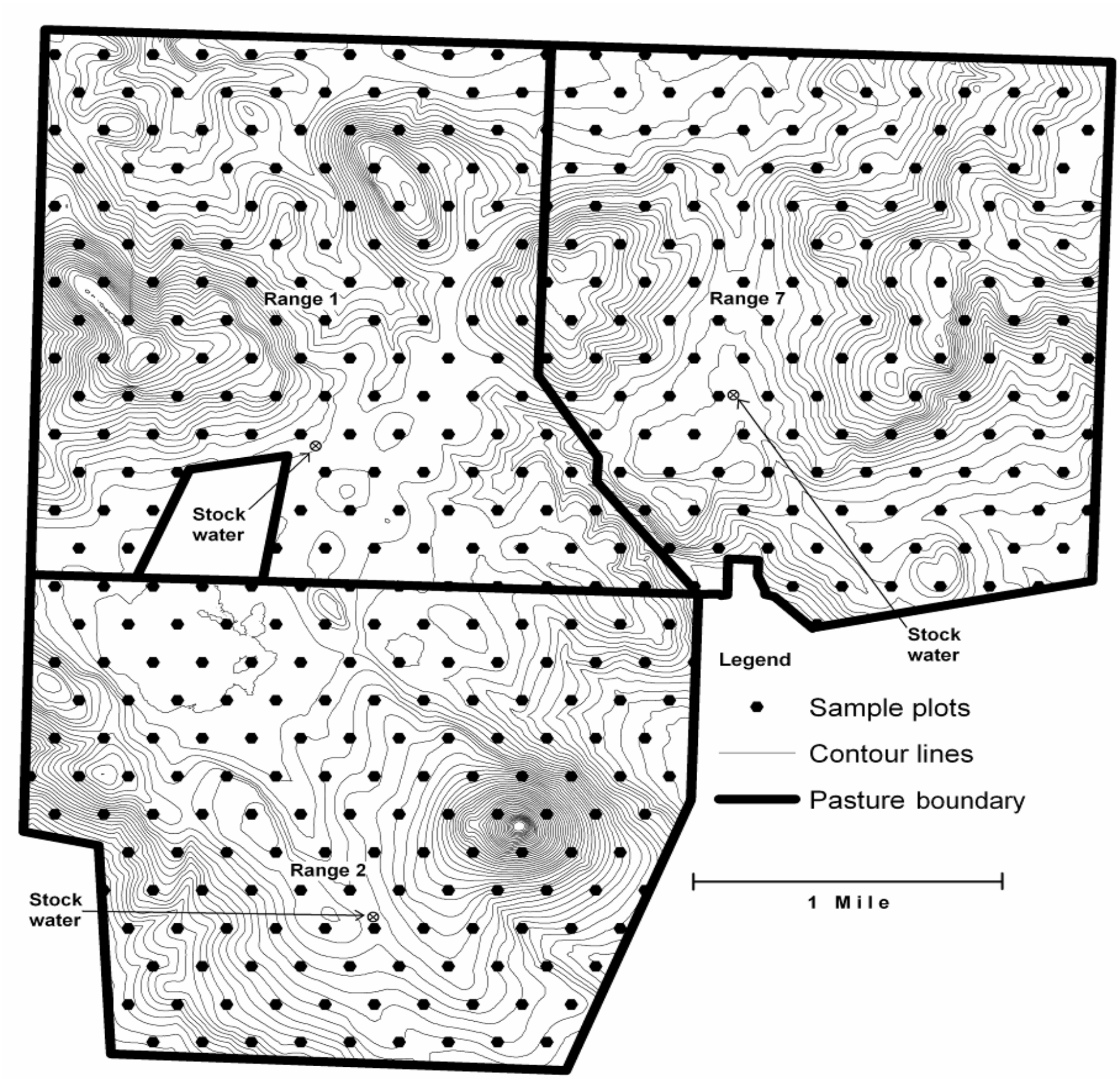


Figure 1. Twenty foot contour lines and positions of 453 forage sampling locations used to assess patterns of forage quantity/quality and the distribution of grazing cattle on the Northern Great Basin Experimental Range near Burns, Oregon in June 2004.

of the three pastures. Four cows in each group were equipped with GPS collars to monitor their whereabouts and establish their activities at 5-minute intervals (Fig. 2).

Yield and nutritional data were entered into a GIS system and maps rendered for each of our forage quantity/quality attributes. These depicted the distribution of forage characteristics across the pastures. The locations of grazing cattle were subsequently overlaid on these same images, and the GIS system asked to furnish the forage quantity/quality composition for the entire pasture and for each location used by cattle (Figure 3).

Figure 2. A global positioning system (GPS) collar used to track movements and activities of cattle in a study evaluating patterns of forage quantity/quality and the distribution of grazing cattle on the Northern Great Basin Experimental Range near Burns, Oregon in June 2004.

RESULTS AND DISCUSSION

Over the 15-day trial, grazing cattle accessed only about 25% of their total pasture area (Fig. 4). Cattle preferred areas with higher than average crude protein and forage digestibility and lower than average standing crop and neutral detergent fiber. The crude protein composition of the pastures and the areas frequented by cattle are depicted in Figure 4. Crude protein of forage across the pastures averaged about 8.8% and 64% of grazing cattle observations occurred in areas exceeding that value. Our initial thoughts were that cattle would seek out the absolute highest quality areas in the pastures, but that did not occur. We speculate that because the highest crude protein areas make up less that 5% of the landscape, it is not efficient for cattle to expend the energy needed to find those limited resources. With about 35% of the area in each pasture furnishing more than adequate forage quality, there was again little physiologic need to search for exceptional herbage. Results for forage digestibility analyses painted about the same picture as the crude protein data.

Cattle also favored portions of the pasture supporting lower than average standing crop (Fig. 5). Classical studies where forage quality is constant across a pasture suggest cattle will seek areas where their forage intake rate can be maximized. That is typically among areas supporting the highest standing crop. We found about 60% of our grazing cattle observations in areas that supported less than average standing crop (318 lbs acre^{-1}).

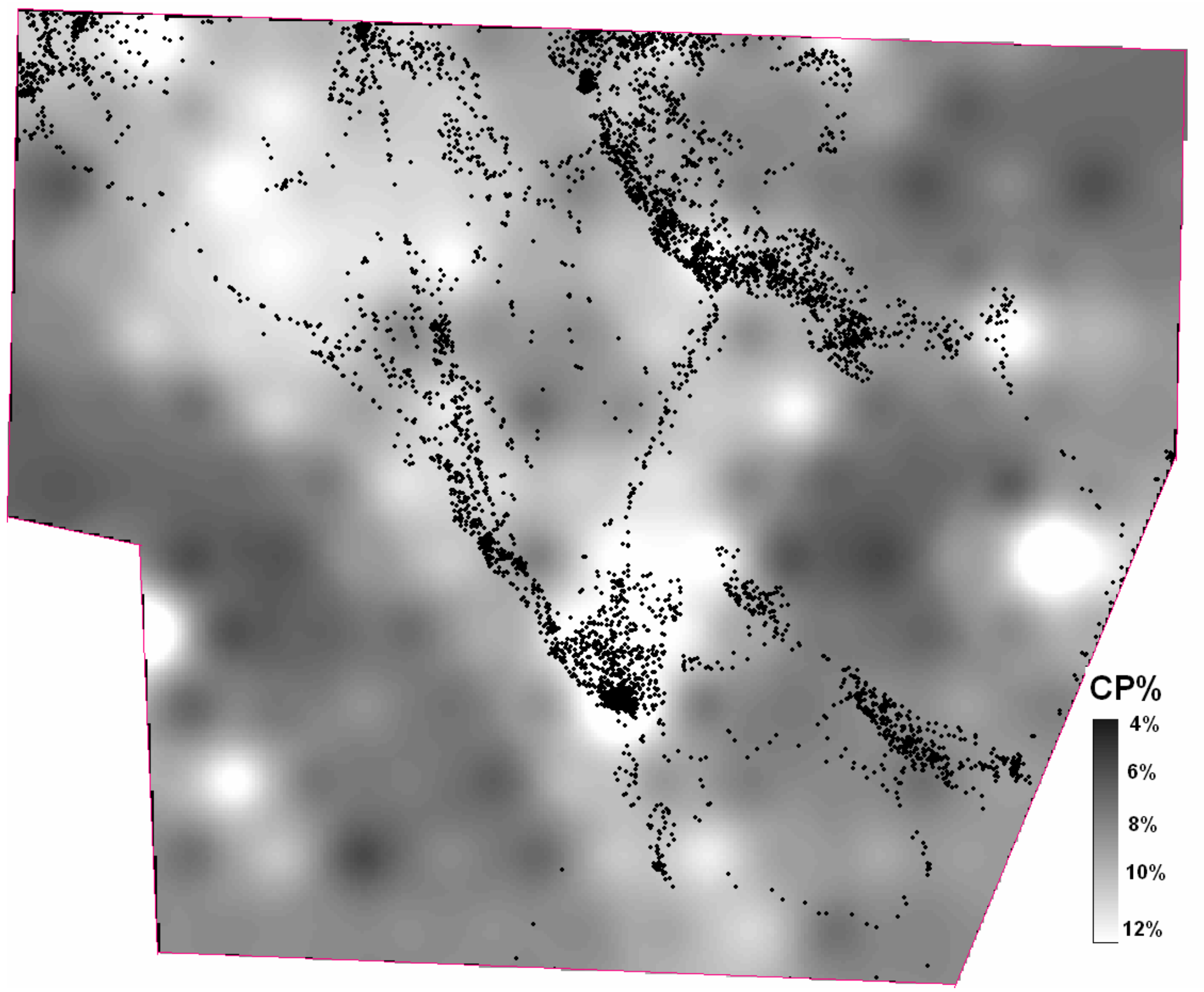


Figure 3. A map depicting the crude protein content of forage in Range 2 and the locations used by grazing cattle in a study evaluating forage quality patterns and the distribution of cattle across the landscape in June 2004 on the Northern Great Basin Experimental Range near Burns, Oregon.

Earlier studies at the experiment station have shown that cattle are very sensitive to and intolerant of standing dead straw intermingled with green herbage. We speculate that cattle were avoiding our most productive sites because they also supported an abundance of "wolfy" forage. Consistent with our standing crop findings, cattle also favored locations supporting low neutral detergent fiber (data not shown). High levels of fiber in forages are associated with reduced digestibility and intake by herbivores, so we would expect cattle to avoid high fiber sites.

CONCLUSIONS

Consistent with optimum foraging theory, grazing cattle most frequently used portions of pastures exhibiting higher than average crude protein and forage digestibility and lower than average standing crop and neutral detergent fiber. We speculate such preferences may explain the seasonal movements we see by cattle in large pastures as the growing and grazing seasons advance. Fidelity to historic grazing patterns, distribution shifts to recently burned locales, seasonal moves to riparian sites, and late-summer movements of cattle to sites supporting palatable shrubs are likely all responses to changing landscape nutritional dynamics. Management practices that alter the nutritional character of the landscape like prescribed burns, fertilization, mowing, controlled grazing, or strategic supplement placement will likely be successful at altering grazing distribution and should be useful management tools for attracting cattle to historically unused locales.

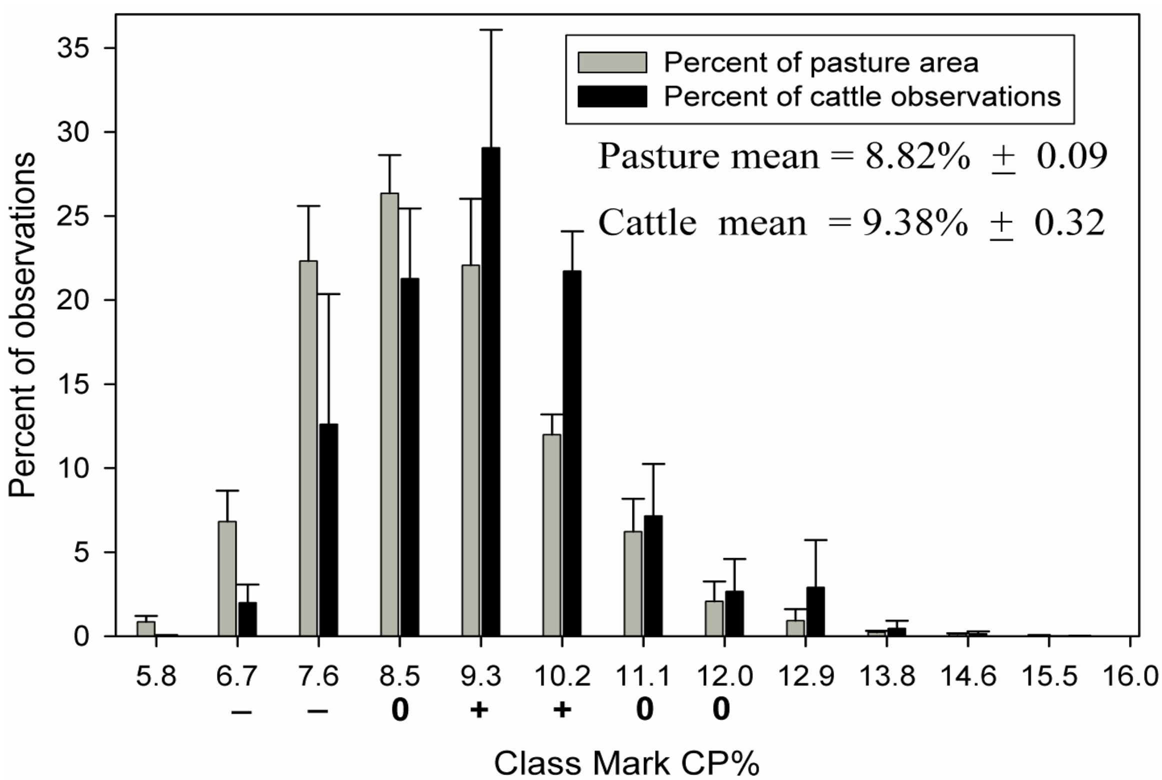


Figure 4. Percentages of the landscape and grazing cattle observed in 13 crude protein classes sampled among 3 pastures on the Northern Great Basin Experimental Range in June 2004 during a study assessing landscape nutritional patterns and their effects on cattle distribution. A "-"beneath the X axis indicates a class was avoided by cattle. A "0" implies a class was used by cattle roughly in proportion to its presence on the landscape, and a "+" designation suggests the class was preferred by cattle. Classes with no symbols were not included in analyses, because they were minor components of the landscape and rarely visited.

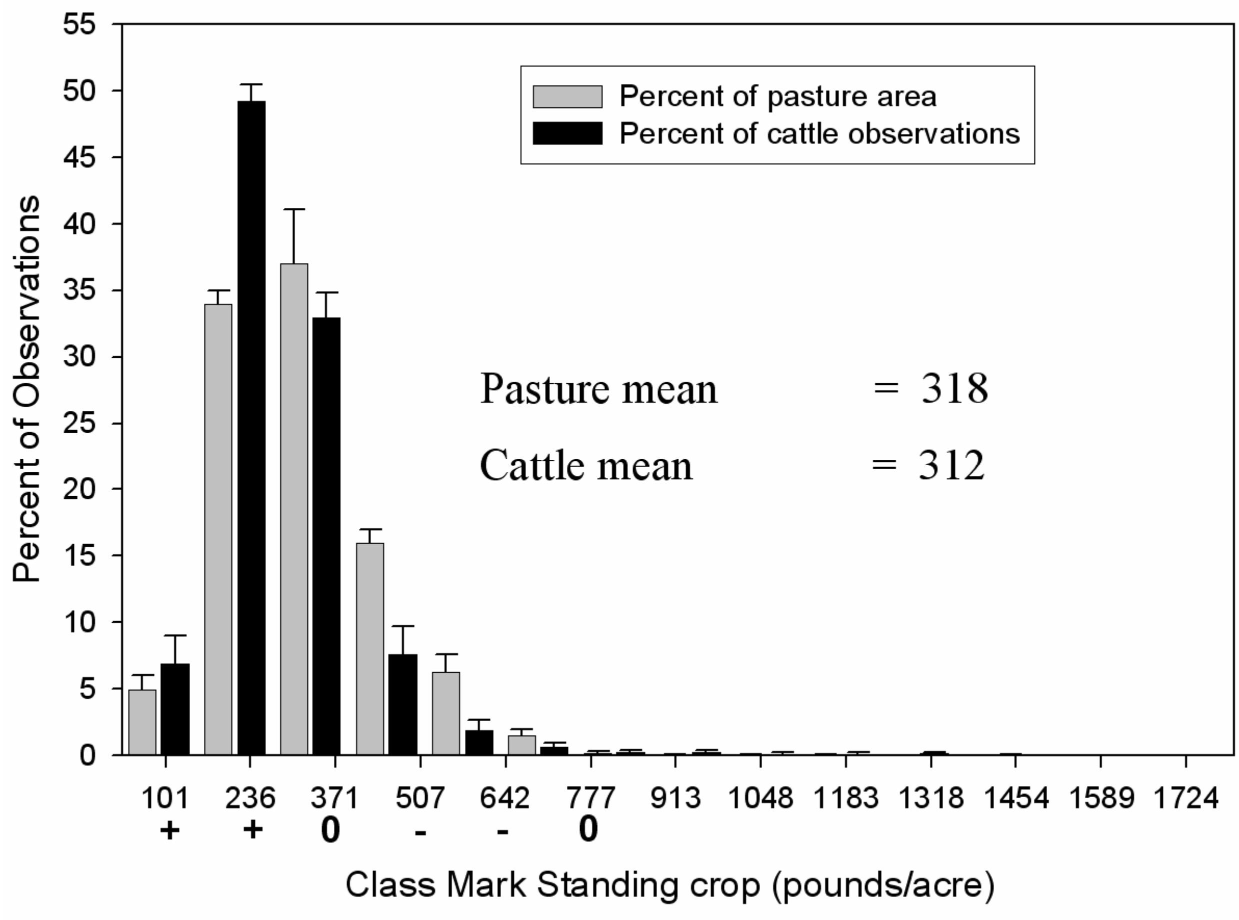


Figure 5. Percentages of the landscape and grazing cattle observed in 13 standing crop classes sampled among 3 pastures on the Northern Great Basin Experimental Range in June 2004 during a study assessing landscape nutritional patterns and their effects on cattle distribution. A "-"beneath the X axis indicates a class was avoided by cattle. A "0" implies a class was used by cattle roughly in proportion to its presence on the landscape, and a "+" designation suggests the class was preferred by cattle. Classes with no symbols were not included in analyses, because they were minor components of the landscape and rarely visited.

Medusahead Ecology and Management

Medusahead Establishment and Dispersal in Sagebrush-Bunchgrass Communities

Kirk W. Davies

SUMMARY

Medusahead is an invasive annual grass that reduces biodiversity and production of rangelands. To prevent medusahead invasion land managers need to know more about its invasion process. Specifically, 1) the timing and spatial extent of medusahead seed dispersal and 2) the establishment rates and interactions with plant communities being invaded. Medusahead seeds dispersed between July and October and did not disperse more than 6.6 feet from their source, without human or animal transport, suggesting that relatively narrow containment barriers around medusahead infestations may be sufficient to significantly slow spread. The ability of medusahead to establish in a plant community was negatively correlated to large perennial grass density. Thus, maintaining large perennial grass is critical to preventing medusahead invasion and increasing large perennial grass density should reduce the susceptibility of a site to medusahead invasion.

INTRODUCTION

Medusahead is an exotic annual grass invading rangelands in the western United States. Its rapid spread into previously uninfested areas is a serious management concern as medusahead invasion has reduced the grazing capacity of rangelands by as much as 90%. Medusahead litter also has a slow decomposition rate which allows it to build up over time and suppress native plants. The build up of medusahead litter also increases the amount and continuity of fine fuel, thus increasing fire frequency to the detriment of native vegetation. The result is often a loss of native species and dense monocultures of medusahead. To prevent medusahead invasion land managers need to know more about its invasion process. Specifically, 1) the timing and spatial extent of medusahead seed dispersal and 2) the establishment rates and interactions with plant communities being invaded.

METHODS

The study was conducted in the northwest foothills of Steens Mountain in southeastern Oregon. Medusahead dispersal was measured using sticky seed traps and medusahead establishment was measured at 12 sites. Vegetation cover and density by species were measured at the 12 sites to determine their impact on medusahead establishment. Wyoming big sagebrush is the dominant shrub and bluebunch wheatgrass or squirreltail is the dominant large perennial grass depending on site.

RESULTS

Most medusahead seeds (449 ± 90 seeds·11 ft^{-2}) dispersed less than 1.6 ft from the invasion front (Fig 1.). From 1.6 to 6.6 ft As distances from the invasion front increased medusahead seed dispersal decreased: at 1.6 ft away, 148 ± 42 seeds·11 ft^{-2} were collected, and at 6.6. ft away 11 ± 6.3 seeds·11 ft^{-2} were collected. No seeds were collected beyond 6.6 ft. Medusahead seeds dispersed from the parent plants from early July (2.3 ± 0.6 seeds/transect) to the end of October (2.3 ± 0.8 seeds/transect). More seeds were trapped in August than the other months (20 ± 4.9 seeds/transect).

Medusahead density was negatively correlated to large perennial grass density and positively correlated to annual grass density of the preexisting plant communities (Fig. 2). These correlations explain 82% of the variation in medusahead density (R^2 = 0.82). Medusahead density was not correlated to density or cover of Sandberg bluegrass and forb functional groups, or to bare ground and litter values. Medusahead cover was also negatively correlated with large perennial grass density (R^2 = 0.44).

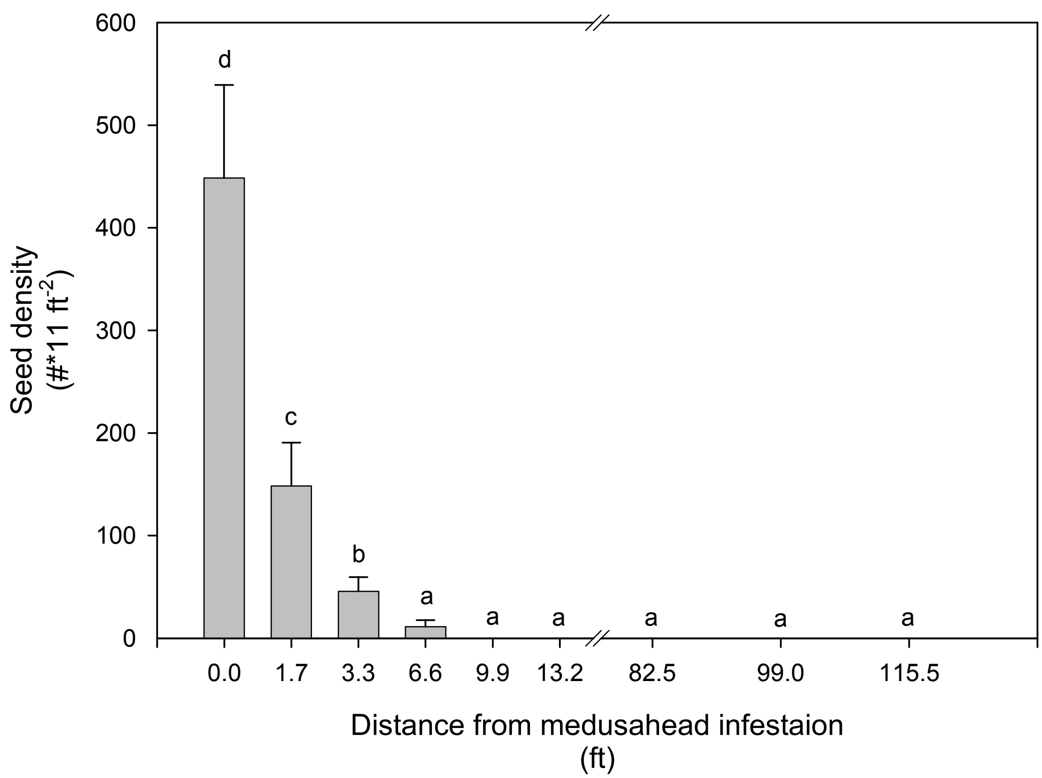


Figure 1. Medusahead seed density (mean + S.E.) trapped at different distances from the medusahead invasion summed across sampling dates, Steens Mountain, Oregon. Different lower case letters indicate differences among distances (P < 0.05).

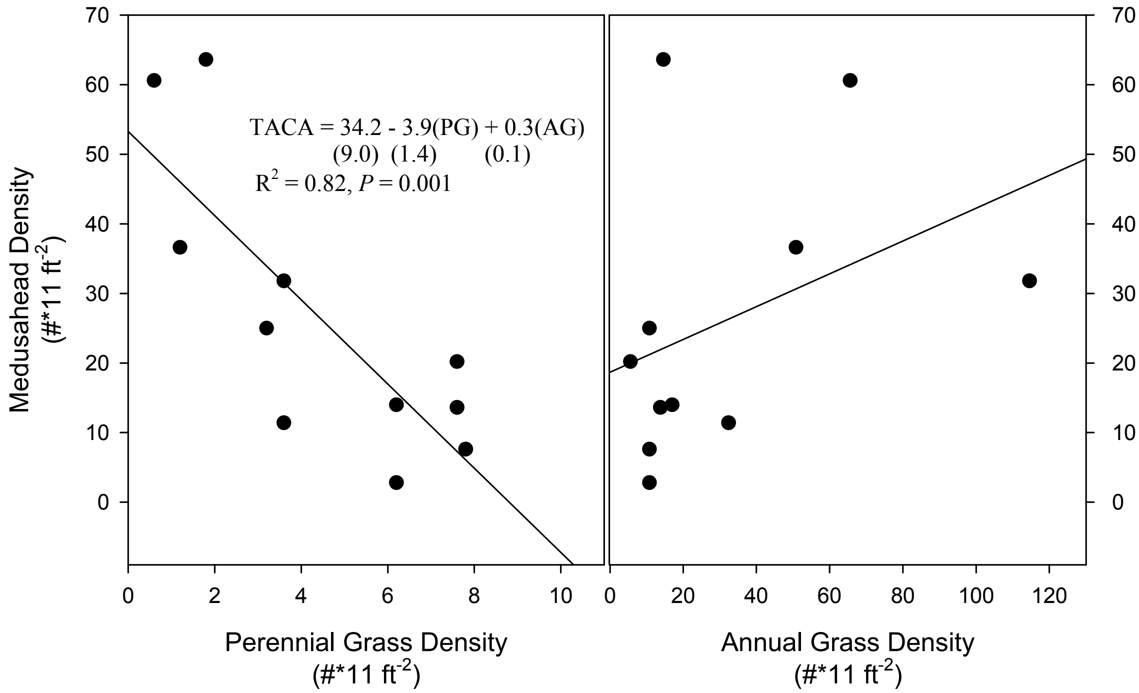


Figure 2. Scatter plot of medusahead density across large perennial grass and annual grass densities of the preexisting plant community with regression lines, Steens Mountain Oregon. TACA8 = medusahead, PG = large perennial grass, and AG = annual grass.

MANAGEMENT IMPLICATIONS

Livestock and vehicle traffic should probably be removed from medusahead infested sagebrush-steppe rangelands during the period when seeds are dispersed to reduce the spread of medusahead. Site differences and between-year variability in precipitation can be expected to influence medusahead seed development and dispersal. Thus, livestock and vehicle exclusion from infestations may vary from year-to-year and site-to-site. The relatively long period of medusahead seed dispersal from July to October may be an adaptation to increase the likelihood of adhesion to animals. Containment zones around infestations can probably be relatively narrow; however, correctly identifying the edge of the infestation is critical. Many medusahead infestations have a diffuse boundary that requires careful scrutiny to identify. Systematic searching for and eradication of new satellite populations will still be necessary to successfully contain medusahead infestations. More research is needed to quantify the dispersal of medusahead by vehicles and animals, especially when soils are sticky due to moisture accumulation. Large perennial grasses appear to be a critical component of sagebrush rangelands that are resistant to medusahead invasion. Promoting and maintaining large perennial grass should be a management priority on rangelands susceptible to medusahead invasion. Large perennial grass and annual grass density may be useful indicators of site susceptibility to medusahead invasion.

REFERENCES

Bovey, R.W., D. Le Tourneau, and L.C. Erickson. 1961. The chemical composition of medusahead and downy brome. Weeds 9:307-311.

Dahl, B.E., and E.W. Tisdale. 1975. Environmental factors relating to medusahead distribution. Journal of Range Management 28:463-468.

Davies, K.W., and R.L. Sheley. 2007. A conceptual framework for preventing the spatial dispersal of invasive plants. Weed Science 55:178-185.

George, M.R. 1992. Ecology and management of medusahead. Davis, CA: University of California Range Science Report 23:1-3.

Hironaka, M. 1961. The relative rate of root development of cheatgrass and medusahead. Journal of Range Management 14:263-267.

Monaco, T.A., T.M. Osmond, and S.A. Dewey. 2005. Medusahead control with fall- and spring-applied herbicides in northern Utah foothills. Weed Technology 19:653-658.

Torell, P.J., L.C. Erickson, and R.H. Haas. 1961. The medusahead problem in Idaho. Weeds 9:124-131.

Young, J.A. 1992. Ecology and management of medusahead (*Taeniatherum caput-medusae* ssp. *asperum* [SIMK.] Melderis). Great Basin Naturalist 52:245-252.

Medusahead Outperforms Squirreltail

Jane Mangold and Kert Young

SUMMARY

Understanding the ecological processes fostering invasion and dominance by medusahead is central to its management. The objectives of this study were 1) to quantify and compare interference between medusahead and squirreltail under different concentrations of soil nitrogen (N) and phoshorus(P) and 2) to compare growth rates of medusahead and squirreltail. We grew medusahead and squirreltail in an addition series in a greenhouse and applied one of four nutrient treatments weekly: 1) low N- low phosphorus (P) (no N or P added), 2) low N-high P, 3) high N-low P, and 4) high N-high P. After 70 days density and biomass by species were sampled. We also grew individual medusahead and squirreltail plants in control soil conditions. Biomass, leaf area, and root length were determined for each species at 14-day intervals over 72 days. Regression models for medusahead and squirreltail suggested N appeared to be playing a much larger role than P in interference between the species. The high N treatment did not increase medusahead's interference ability relative to squirreltail as we had hypothesized. Medusahead typically imposed a two to seven time's stronger influence on interference relationships than squirreltail. Medusahead accumulated biomass, leaf area, and root length twice as fast as squirreltail. Results from our study suggest that medusahead seedlings will dominate over squirreltail seedlings. To restore squirreltail to medusahead-infested rangeland, medusahead densities should be reduced with integrated weed management strategies. On medusahead-free rangeland, prevention, early detection, and eradication programs are critical.

INTRODUCTION

Medusahead (*Taeniatherum caput-medusae* (L.) Nevski) is a nonindigenous, invasive winter-annual grass that threatens rangeland systems in the intermountain West. Medusahead is estimated to occupy 2 million ha. in the Great Basin where it displaces native vegetation and forms exclusive stands. It may disrupt nutrient, water, and fire cycles and is almost worthless forage for livestock. When present, medusahead may reduce forage production up to 80%.

Rangeland dominated by medusahead is often devoid of competitive desirable plants. In such cases, introducing and establishing competitive plants is essential for successful management of infestations and the restoration of desirable plant communities. Squirreltail (*Elymus elymoides* (Rafin.) Swezey) has been identified as a potential species for restoration of medusahead-infested range and wild land and has been observed to establish in medusahead stands over time. An early- to mid-seral native bunchgrass common to western rangeland, it germinates across a range of soil temperatures and its cool season root growth may help explain its ability to compete with annual grasses. A variety of other attributes may help squirreltail compete with medusahead including self-pollination, wide ecotypic variation, and efficient seed dispersal mechanisms.

We conducted an addition series study that evaluated the effects of nitrogen (N) and phosphorus (P) additions on the interference between medusahead and squirreltail. Another study compared growth rates of the two species. The overall objectives were 1) to quantify and compare interference between medusahead and squirreltail under different concentrations of soil N and P, and 2) to compare growth rates of medusahead and squirreltail under soil N and P availabilities found in field soil collected locally. We hypothesized that 1) N and P additions would increase medusahead's interference ability relative to the native grass in the interference study and 2) medusahead would display higher growth rates and biomass accumulation than squirreltail.

METHODS

Two studies were simultaneously conducted in a greenhouse at the Eastern Oregon Agricultural Research Center in Burns, Oregon, from May to August, 2005. Medusahead seed was collected locally and squirreltail seed was purchased from L and H Seed in southeastern Washington. Polyvinyl chloride pipe was used to construct growth tubes 0.5 m deep for the interference study and 1.0 m deep for the growth analysis study. Soil that had supported squirreltail and medusahead was collected from two sites near John Day, Oregon, and sieved through to remove rocks and large roots. Soil was mixed with concrete-grade sand and placed in the growth tubes. Seeds of both species were uniformly scattered across the surface of each tube, depending on the experiment, and covered with approximately 2 mm of field soil.

Interference study

Medusahead and squirreltail were planted into the prepared growth tubes in an addition series design. Five seed density levels of medusahead (0, 1, 5, 25, 125 pure live seeds per pot) were fully mixed with the same five density levels for squirreltail seed for a total of 25 density combinations. Each grouping of the 25 density combinations received one of four nutrient treatments and was replicated three times in each of two separate trials. Each trial lasted approximately 70 days.

Each planting matrix received one of four nutrient treatments weekly: 1) low N low P (LN-LP) was the control with no N or P added to the pots; 2) low N high P (LN-HP) added 250 ml of a 600 µM P solution in the form of calcium phosphate; 3) high N low P treatment (HN-LP) added 250 ml of an 8,400 µM N solution in the forms of calcium nitrate and potassium nitrate; and 4) high N high P treatment (HN-HP) added 250 ml of an 8,400 µM N and 600 µM P solution in the forms of calcium nitrate, potassium nitrate, and potassium phosphate. Growth tubes were misted twice daily as needed throughout the study to prevent water stress. After 70 days, density per growth tube of each species was counted and aboveground biomass clipped approximately 5 mm above the soil surface. Aboveground biomass was dried for 72 hours at 50°C and weighed.

Data were grouped by treatment for each trial and fit to multiple linear regression. The inverse of medusahead and squirreltail individual aboveground biomass per plant was predicted using medusahead and squirreltail final densities per growth tube as independent variables. Models were of the form:

$$y_m^{-1} = \beta_{m0} + \beta_{mm}N_m + \beta_{ms}N_s \text{ (medusahead)}$$
$$y_s^{-1} = \beta_{s0} + \beta_{ss}N_s + \beta_{sm}N_m \text{ (squirreltail)}$$

where y_m and y_s represent the average aboveground biomass per plant for medusahead and squirreltail, respectively. The regression coefficients β_{m0} and β_{s0} represented the maximum aboveground biomass for a medusahead and squirreltail plant grown in isolation, respectively. A smaller number indicates greater biomass due to the inverse operation. β_{mm} and β_{ss} represent the effect of species density upon its own biomass (intraspecific interference) from the medusahead and squirreltail models, respectively. β_{ms} and β_{sm} represent the effect of the neighboring species density on the mean biomass of the response species (interspecific interference). N_m and N_s represented the density per growth tube of medusahead and squirreltail, respectively. Slopes from the regression models for each nutrient treatment were compared by calculating variance ratios. The relative interference ability for both species under each nutrient treatment was calculated by dividing the intraspecific interference coefficient by the interspecific interference coefficient.

Growth analysis study

In two separate trials, five seeds of medusahead or squirreltail were uniformly scattered across the surface of 40 (20 for each species per trial) prepared growth tubes and covered with approximately 2 mm of field soil. The density of each growth tube was reduced to one vigorous seedling following establishment. Growth tubes were misted twice daily as needed throughout the study to prevent water stress. Each trial lasted approximately 70 days. No nutrient treatments were applied, so N and P levels were equivalent to the LN-LP (control) treatment in the interference study.

Every 14 days, above- and below-ground biomass of four randomly selected squirreltail and four randomly selected medusahead plants were harvested. Above and below ground biomass was separated, leaf area was quantified, and root length was determined. Above and below ground biomass was dried and weighed. Root:shoot ratio was calculated from above- and below-ground biomass measurements.

Data were natural log transformed and fit to a linear regression to estimate the instantaneous growth rate based on total biomass, leaf area, and root length over the 70-day period. Slopes were compared by calculating variance ratios.

RESULTS AND DISCUSSION

Interference Study

For both Trial 1 and Trial 2, all models predicting medusahead or squirreltail aboveground biomass were significant (P<0.01). Regression model coefficients for both species generally differed between the high and low N treatments, but not between the high and low P treatments (Tables 1 and 2). Nitrogen appeared to be playing a much larger role than P in interference between medusahead and squirreltail.

The invasive grass medusahead outperformed the native grass squirreltail in all aspects of interference. High nutrient availability did not increase medusahead's interference ability relative to squirreltail as hypothesized. In the low N treatments for Trial 1, the predicted maximum aboveground biomass for an isolated medusahead plant (β_{m0}) was about 0.3 g (1/3.16 for low P and 1/3.47 for high P), while the high N treatments resulted in a nonsignificant regression coefficient for predicted aboveground biomass. Squirreltail predicted maximum aboveground biomass for an isolated individual (β_{s0}) ranged from about 0.07 g (1/15.02) to 0.15 g (1/6.71 and 1/6.81) in the low N treatments across trials. A nonsignificant regression coefficient suggested the maximum biomass of an individual would be very large because of the reciprocal operation (i.e., 1/≈0). In all cases, the predicted maximum aboveground biomass for a medusahead plant was greater than that of a squirreltail plant.

In the medusahead models intraspecific interference was more intense than interspecific interference, while in the squirreltail models interspecific interference was more intense than intraspecific interference. Intraspecific interference coefficients for medusahead (β_{mm}) decreased from approximately 0.35 with low N treatments to 0.1 with high N treatments in Trial 1 and 0.30 to 0.11 in Trial 2 (Table 1). In the low N treatments in Trial 1, the interspecific interference coefficient (β_{ms}) was not significant, suggesting squirreltail density

Table 1. Trial 1 and Trial 2 multiple linear regression models with medusahead and squirreltail growth tube density predicting the inverse of individual medusahead biomass (g plant^{-1}).

Treatment	β_{m0}	β_{mm}	β_{ms}	β_{mm}/β_{ms}	r^2
			Trial 1		
loNloP	3.16 (1.25)	0.34 (0.02)	0.06 (NS)	3.4×10^3 (a)	0.86
hiNloP	0.33 (NS)	0.11 (0.00)	0.04 (0.01)	2.57 (b)	0.96
loNhiP	3.47 (1.52)	0.37 (0.02)	0.04 (NS)	3.7×10^3 (a)	0.83
hiNhiP	0.16 (NS)	0.11 (0.00)	0.06 (0.01)	1.92 (b)	0.97
			Trial 2		
loNloP	1.39 (NS)	0.29 (0.01)	0.06 (0.02)	4.74 (a)	0.92
hiNloP	-0.03 (NS)	0.11 (0.00)	0.04 (0.00)	2.87 (b)	0.98
loNhiP	1.30 (NS)	0.31 (0.01)	0.04 (0.02)	7.05 (a)	0.89
hiNhiP	0.03 (NS)	0.11 (0.00)	0.04 (0.00)	2.98 (b)	0.96

β_{m0}=inverse mean biomass of an individual medusahead plant grown in isolation, β_{mm}=effect of medusahead density on medusahead biomass per plant, β_{ms}=effect of squirreltail density on medusahead biomass per plant. β_{mm}/β_{ms}=relative interference ratio of the two species, and r^2=coefficient of determination. Numbers in parentheses are standard errors for coefficients significantly different from zero (P=0.05). NS=not significant. Slopes of models with different letters in parentheses in the relative interference ratio column are statistically different from one another.

had no effect on medusahead biomass. Interspecific interference increased to about 0.05 in the high N treatments in Trial 1 and in all treatments during Trial 2. Squirreltail intraspecific interference coefficients (β_{ss}) decreased from low N treatments to high N treatments with values ranging from 0.41 to 0.10, respectively (Table 2). Interspecific interference coefficients (β_{sm}) ranged from 0.40 to 0.76 and were generally higher in high N treatments.

All of the relative interference ratios (β_{mm}/β_{ms}) for medusahead were greater than one, especially in the high N treatments, while the opposite was true for squirreltail ($\beta_{ss}/\beta_{sm} < 1$) (Tables 1 and 2). This suggests medusahead was imposing more interference on both its own biomass and on squirreltail biomass than was squirreltail on medusahead biomass or its own biomass. The effect of medusahead density on medusahead biomass was generally about 2 to 7 times greater than the effect of squirreltail density on medusahead biomass. The effect of squirreltail density on its own biomass was generally about 25-80% of the effect of medusahead density on squirreltail biomass.

Table 2. Trial 1 and Trial 2 multiple linear regression models with medusahead and squirreltail growth tube density predicting the inverse of individual squirreltail biomass (g plant^{-1}).

Treatment	β_{s0}	β_{ss}	β_{sm}	β_{ss}/β_{sm}	r^2
			Trial 1		
loNloP	15.02 (5.30)	0.41 (0.13)	0.52 (0.10)	0.80 (a)	0.43
hiNloP	1.89 (NS)	0.12 (0.04)	0.43 (0.03)	0.26 (b)	0.82
loNhiP	16.13 (NS)	0.42 (0.20)	0.76 (0.14)	0.55 (a)	0.44
hiNhiP	3.61 (NS)	0.10 (NS)	0.40 (0.05)	2.5×10^{-4} (b)	0.62
			Trial 2		
loNloP	6.71 (2.86)	0.33 (0.05)	0.44 (0.04)	0.75 (a)	0.73
hiNloP	3.62 (NS)	0.12 (0.04)	0.26 (0.03)	0.45 (b)	0.60
loNhiP	6.81 (2.68)	0.31 (0.05)	0.58 (0.04)	0.54 (b)	0.83
hiNhiP	0.92 (NS)	0.18 (0.03)	0.24 (0.02)	0.73 (b)	0.74

β_{s0}=inverse mean biomass of an individual squirreltail plant grown in isolation, β_{ss}=effect of squirreltail density on squirreltail biomass per plant, β_{sm}=effect of medusahead density on squirreltail biomass per plant. β_{ss}/β_{sm}=relative interference ratio of the two species and r^2=coefficient of determination. Numbers in parentheses are standard errors for coefficients significantly different from zero (P=0.05). NS=not significant. Slopes of models with different letters in parentheses in the relative interference ratio column are statistically different from one another.

Growth Analysis Study

Data collected from each trial were analyzed separately because biomass accumulation was greater in Trial 2. Only results from Trial 2 are presented graphically (Fig. 1), but results were similar across trials.

Consistent with our second hypothesis, we found that medusahead grew bigger and faster than squirreltail. Invasive and/or annual species commonly display high growth rates and large biomass accumulation. Medusahead growth rate was higher than squirreltail's in both trials. Medusahead total biomass increased by 0.11 g/day while squirreltail total biomass increased by 0.05 g/day during Trial 2 (Fig. 1a). During Trial 1 medusahead total biomass increased by 0.05 g/day compared to a 0.02 g/day increase for squirreltail. A high growth rate is one mechanism that may be critical to medusahead's success. A species with a higher growth rate may dominate because it can establish faster, increase in size more quickly, and gain more access to resources than a slower growing species.

Medusahead generally accumulated leaf area and root length at a faster rate than squirreltail in both trials. Medusahead leaf area increased by about 4.5 cm^2/day and 13.3 cm^2/day in Trial 1 and Trial 2, respectively (Figure 1b); squirreltail leaf area increased by about 4.9 and 2.3 cm^2/day for Trial 1 and Trial 2, respectively (Figure 1b). Root length accumulation showed similar trends with medusahead root length increasing about 2.5 (Trial 1) to 4 (Trial 2) times faster than squirreltail root length (Figure 1c). However, squirreltail root:shoot ratios based on biomass were approximately 1.5 times greater than medusahead root:shoot ratios in Trial 2 at 30, 44, and 58 days post planting.

The growth analysis study supported results of the interference study and offers important insight into why medusahead appeared to impose more interference than squirreltail. When resources such as N are plentiful, the species that is most capable of growing rapidly and exploiting available resources, in this case medusahead, will benefit the most. Medusahead may be outperforming squirreltail via resource (i.e. N) preemption because 1) medusahead displayed faster growth rates and greater increases in leaf area and root length than squirreltail and 2) medusahead individual plant weight (i.e. total biomass) was always greater than that of squirreltail. Furthermore, the influence of medusahead on its own biomass was more intense than the influence of squirreltail on medusahead's biomass, and this intensity was lessened when N was increased.

Squirreltail has been observed to establish in medusahead stands over time, which is contradictory to what our results might predict. One reason for this discrepancy may be because squirreltail is highly ecotypic. Ecotypic variation may provide genotypes that resist medusahead invasion more so than others. Yet another reason may be because our study quantified seedling-seedling interference instead of mature squirreltail-medusahead seedling interference. The perennial nature of squirreltail and its tendency to have higher root to shoot ratios, as indicated in the growth analysis, may confer some advantages over time that we were not able to distinguish in this study.

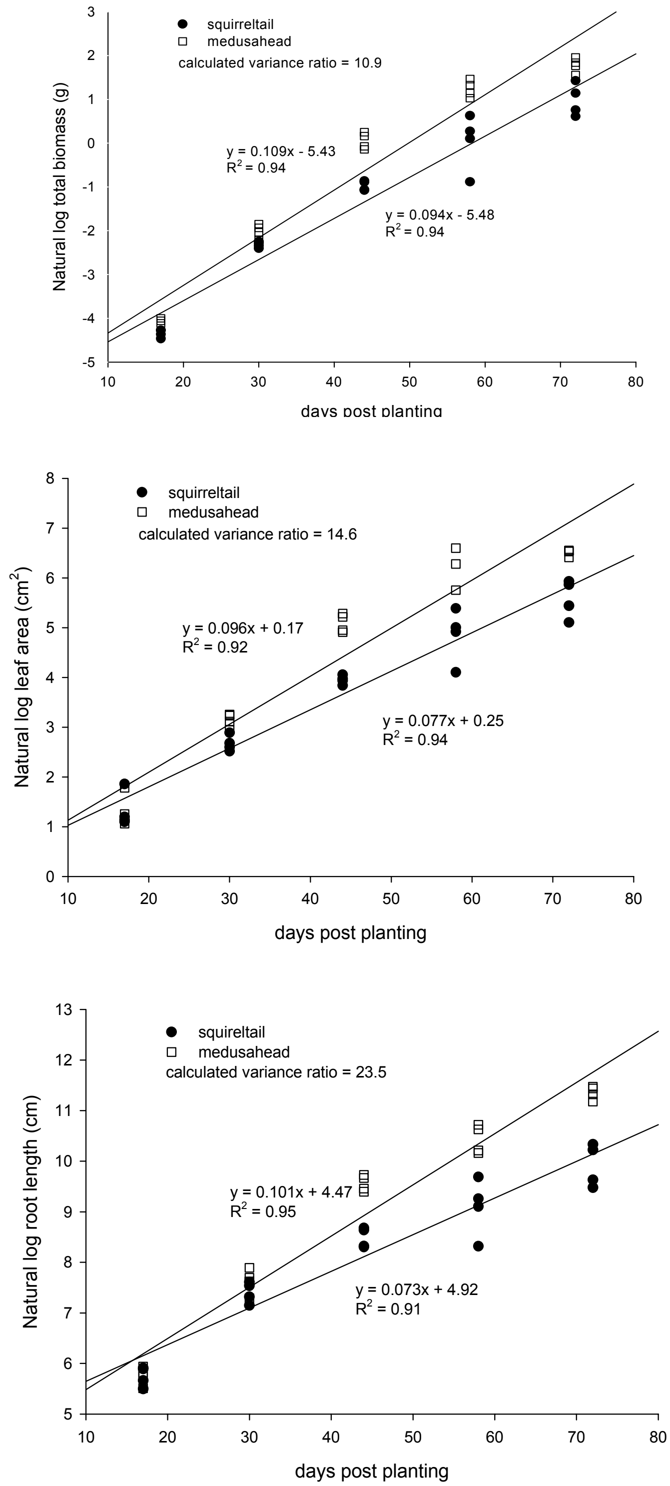


Figure 1. Growth rate (a), leaf area increase (b), and root length increase (c) of isolated individuals of squirreltail and medusahead for Trial 2. Data were linearized by taking the natural log of measured total biomass, leaf area, and root lengths to allow comparison of slopes. Critical value of variance ratio used to compare slopes$_{(\alpha = 0.05)}$ = 6.3. Calculated variance ratio > critical value suggests slopes are different.

IMPLICATIONS

The results from our study suggest that squirreltail is not likely to effectively compete with medusahead in the seedling stage. Therefore, in order to restore squirreltail to rangeland dominated by medusahead, densities of medusahead seed in the seed bank should be reduced by carefully timing various integrated weed management strategies like burning, herbicide use, and grazing. Seeding squirreltail at high rates may also improve establishment success. Once established, squirreltail may be able to maintain itself through perennial resource allocation patterns, but would not likely eradicate medusahead. We suspect that revegetation of medusahead infested rangeland will require large quantities of resources and time. Aggressive and comprehensive prevention programs that include early detection and eradication are critical for conserving rangeland that is currently relatively medusahead-free.

This paper is an abridged version of Young, K. and J. Mangold. 2008. Medusahead (*Taeniatherum caput-medusae*) outperforms squirreltail (*Elymus elymoides*) through interference and growth rate. Invasive Plant Science and Management 1:73-81.

REFERENCES

Hardegree, S.P., T. A. Jones, and S. S. Van Vactor. 2002. Variability in thermal response of primed and non-primed seeds of squirreltail [*Elymus elymoides* (Raf.) Swezey and *Elymus multisetus* (J. G. Smith) M. E. Jones]. Annals of Botany 89:311-319.

Hironaka, M. 1961. The relative rate of root development of cheatgrass and medusahead. Journal of Range Management 14:263-267.

Hironaka, M. and B.W. Sindelar. 1973. Reproductive success of squirreltail in medusahead infested ranges. Journal of Range Management 26:219-221.

Jones, T.A. 1998. Viewpoint: the present status and future prospects of squirreltail research. Journal of Range Management 51:326-331.

Jones, T.A., D.C. Nielson, J. T. Arrendondo, and M. G. Redinbaugh. 2003. Characterization of diversity among 3 squirreltail taxa. Journal of Range Management 56:474-482.

Miller, H.C., D. Clausnitzer, and M. M. Borman. 1999. Medusahead. Pages 271-281 in R. L. Sheley and J. K. Petroff, eds. Biology and management of noxious rangeland weeds. Corvallis, OR: Oregon State University Press.

Young, J. A. 1992. Ecology and management of medusahead (*Taeniatherum caput-medusae ssp. Asperum* [Simk.] Melderis). Great Basin Naturalist 52:245-252.

Effects of Imazapic on Target and Nontarget Vegetation During Revegetation

Roger L. Sheley, Michael F. Carpinelli, Kimberly J. Reever Morghan, And Edward A. Vasquez

SUMMARY

Medusahead is an introduced, winter-annual grass covering millions of acres of western rangelands. It forms large monocultures and has a dense thatch cover that resists the establishment of desirable vegetation. Prescribed fire can remove medusahead litter and improve plant establishment. Medusahead control is fundamental to establishing desirable vegetation that will, in turn, resist future invasion. Imazapic is an effective herbicide for control of medusahead, but more information is needed on its effects on desirable vegetation. Therefore, existing medusahead infestations were burned at two different sites in June 2003 to test how Imazapic application rate and timing affected medusahead, desirable seeded plant species, and other nontarget vegetation on burned and unburned rangeland in southeastern Oregon. Following the burn, imazapic was applied at seven increasingly concentrated rates between July and October of 2003 in a randomized strip-plot design field experiment. In November 2003, seven different desirable plant species were drill-seeded separately across the imazapic areas. Data on cover and density of medusahead and seeded plant species were collected in 2004 and 2005. Cover data of nontarget vegetation were collected in the summer of 2005. Medusahead cover was the highest in control plots that did not receive imazapic and lowest in plots that received the highest herbicide rates. Seeded plant species established in the study plots, but their response to herbicide rate showed few consistent patterns. For example, some of the seeded plant species showed little response to herbicide, whereas others appeared to establish best at different herbicides rates, depending on site and whether the plots were burned or unburned. Site and burn treatment also affected how imazapic rate or application month influenced cover of perennial or annual grasses or forbs.

INTRODUCTION AND OBJECTIVES

Medusahead is currently one of the greatest threats to plant communities in the Great Basin. Medusahead is capable of forming monoculture stands that burn readily and resist reestablishment of native vegetation. Medusahead is also largely unpalatable by livestock; thus, invasion results in economic losses to rural communities, and it is reported that medusahead-dominated ranges have suffered a 40 to 70% reduction in grazing capacity. This aggressive, invasive, winter-annual grass native to Eurasia grows in a variety of environments but is restricted to regions with 10 to 40 inches annual precipitation with hot dry summers and cool wet winters. Medusahead commonly occupies clay soils that maintain soil moisture late into the growing season, in arid environments, and in well-developed loam soils where soil moisture is sufficient for the plant to mature.

Some success in controlling medusahead populations has been achieved by herbicide use; however there has been very little published research on the effectiveness of imazapic at controlling medusahead. Previous research suggests that one challenge of managing medusahead with plantings of competitive species is to find a rate of imazapic that will offer control of medusahead without damaging non-target vegetation. More research is needed to understand how medusahead will respond to different rates of imazapic and different herbicide application timings and whether those patterns are the same in burned and unburned fields.

An integrative management strategy that combines herbicides, fire, and reseeding of competitive plant species can create a diverse plant community that will resist future invasions. Perennial grass plantings may provide a plant community better able to resist further invasion of annual weeds such as medusahead, but these plantings may fail if competition from annual invaders is not controlled. The purpose of this study was to evaluate the effects of imazapic rate and timing of application on various grass species and forbs in burned and non-burned pastures.

METHODS

The study was conducted at two sites in eastern Oregon from 2003 to 2005. The first site, Mullin Ranch is located near John Day, Oregon (Grant County, lat 44° 26' 5.05" N, long 118° 56' 18.29" W). The second site, Lamb Ranch, is located near Drewsey, Oregon (Grant County, lat 43° 26' 54.98" N, long 118° 26' 43.94" W). At each site, a 5-acre portion of an existing medusahead infestation was burned in June 2003. Another 5-acre portion was left as an unburned control. On both the burned and unburned areas at each site, imazapic was applied at one of four times and one of seven (0, 1.0, 1.5, 2.0, 2.5, and 3.0 ounces per acre) herbicide application rates. The timing of herbicide application was late-July, late-August, late-September, and late-October 2003, and each application month was replicated seven times. In November 2003, monocultures of seven plant species commonly used in revegetation were drill-seeded perpendicularly across the imazapic treated areas, resulting in three replications of the seeding treatment. These were: (1) thickspike wheatgrass (ELLA), (2) Siberian wheatgrass (AGFR), (2) bluebunch wheatgrass (PSSP), (3) squirreltail (ELEL), (4) Sandberg bluegrass (POSE), (5) winterfat (KRLA), (6) forage kochia (KOPR), and (7) control, unseeded. One row in each of the three replications was left unseeded as a control.

RESULTS

Medusahead Cover

Medusahead cover tended to be highest in the plots that received no imazapic and lowest in plots that received the three highest application rates. Medusahead cover also tended to be higher in the unburned plots (Figs. 1 and 2). However, although imazapic reduced medusahead cover in both years, even the highest concentrations of imazapic did not reduce medusahead cover to zero. Application timing had a small effect on herbicide effectiveness; July applications were slightly less effective than applications in the other 3 months. Mean medusahead cover was lower in burned plots than in unburned plots, and that difference in cover was especially noticeable in plots that received no imazapic or low imazapic concentrations (Figs. 1 and 2).

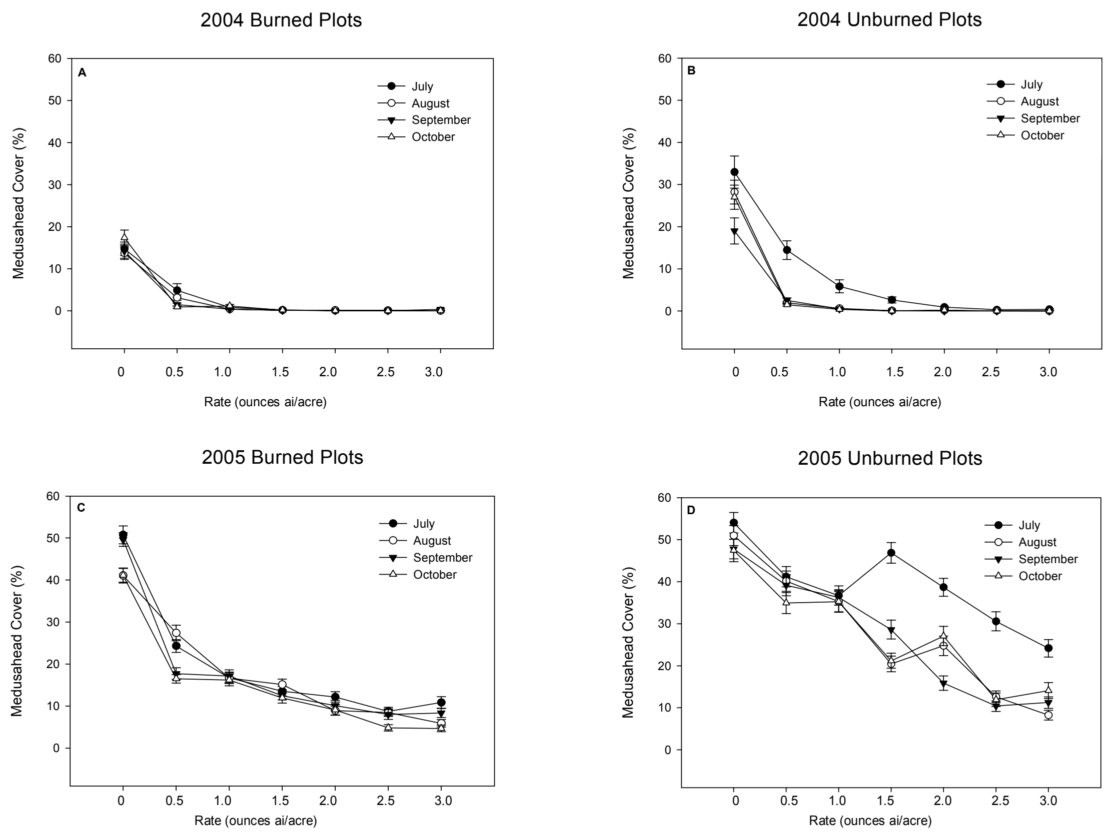


Figure 1. Lamb Ranch (Drewsey, OR) medusahead cover in response to herbicide rate and application month. Values are means ± SE. (A) 2004 burned plots, (B) 2004 unburned plots, (C) 2005 burned plots, and (D) 2005 unburned plots.

Seeded Plant Species Cover and Density

Cover by seeded species was low throughout this study; the average cover was below 3% at Lamb Ranch and under 9% at Mullin Ranch. However, the seeded species were established at these two sites. Perennial grass cover did not show a clear pattern with regard to herbicide rate or herbicide application month. The cover of annual grasses other than medusahead tended to be higher at the lower imazapic application rates, especially in the first year of the study. In addition, no consistent pattern of response by perennial forbs to imazapic application rate was seen, with the exception of a few unburned plots that showed higher coverage by perennial forbs at higher rates. The relationship between annual forb cover and herbicide rate varied greatly between years and showed few consistent patterns. In some of the plots, the relationship of annual forb cover to imazapic rate switched between years; in the first year of the study, annual forb cover was lowest in the plots that received the highest rates of imazapic, but by the second year of the study, annual forb cover was lowest in plots that received the lowest rate of imazapic.

The patterns seen for seeded species density at Lamb Ranch in 2005 were similar to those seen for seeded species cover, and few patterns between density and herbicide rate were visible. As seen for Lamb Ranch, the results for seeded species density at Mullin Ranch in 2005 were similar to those for seeded species cover, with no clear pattern between herbicide rate and density. However, differences between the burned and unburned plots were quite pronounced. The highest density was seen for Siberian wheatgrass, followed by squirreltail, in the burned plots, whereas in the unburned plots, the highest density was seen for Sandberg bluegrass. Seeded perennial grasses appear to have established more successfully than seeded forbs during this study (data not shown). There was a large difference in forb establishment between sites and response to burning, with winterfat only establishing in the unburned plots at Lamb Ranch and not at all at Mullin Ranch. Establishment of all five perennial grass species was higher than forb establishment, and some, such as Siberian wheatgrass and squirreltail, did well on many plots.

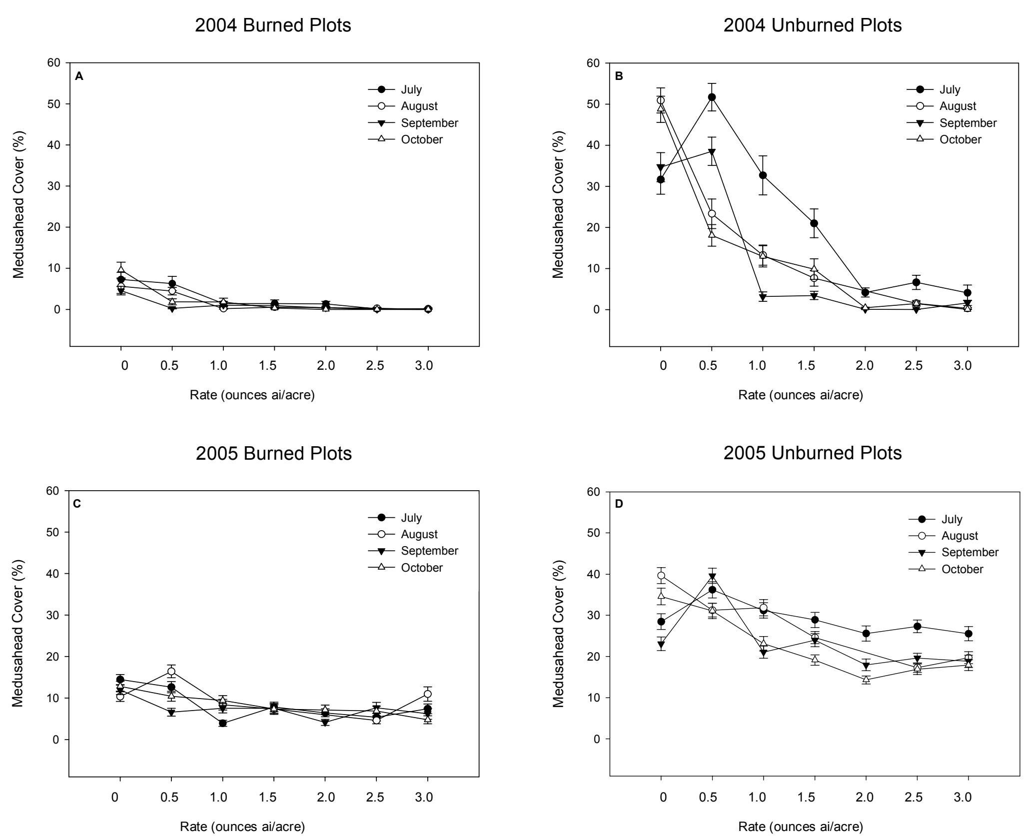


Figure 2. Mullin Ranch (John Day, OR) medusahead cover in response to herbicide rate and application month. Values are means ± SE. (A) 2004 burned plots, (B) 2004 unburned plots, (C) 2005 burned plots, and (D) 2005 unburned plots.

MANAGEMENT IMPLICATIONS

Imazapic application offers effective control of medusahead before seeding with desirable species, and that control is more effective in plots that have been burned to remove medusahead thatch. The results from this study suggest that imazapic alone can control medusahead in the initial year; however, in subsequent years, medusahead is likely to reinfest the area. Other herbicides can also be used to control medusahead, such as glyphosate, atrazine, bromacil, siduron with picloram, and dalapon. Careful selection and application of these herbicides can allow control of medusahead with minimal damage to desired species. Using an integrated management strategy that uses prescribed burns in conjunction with imazapic can potentially result in better control at lower herbicide concentrations, although that control may be of short duration. The earliest application of imazapic in July appears to be less effective than later application. Forbs may be more difficult to establish during medusahead control and revegetation. Imazapic appears to control annual grasses other than medusahead during the first year after application, although that control may only be of short duration as those grasses can sprout readily from seed the year after management. Furthermore, our study found few consistent patterns relating the rate of imazapic to the cover of annual or perennial forbs.

REFERENCES

George, M. R. 1992. Ecology and management of Medusahead. Davis, CA: University of California. Range Science Report 32, 1-3.

Monaco, T. A., T. M. Osmond, and S. A. Dewey. 2005. Medusahead control with fall- and spring-applied herbicides on Northern Utah foothills. Weed Technology 19:653-658.

Young, J. A., R. A. Evans, and J. Robison. 1972. Influence of repeated annual burning on a medusahead community. Journal of Range Management 25:372-375.

Crested Wheatgrass Defoliation Intensity and Season of Use on Medusahead Invasion

Roger L. Sheley, Brett S. Bingham, Tony J. Svejcar, and Edward A. Vasquez

SUMMARY

The objective of this study was to determine the effects of crested wheatgrass defoliation intensity and timing on medusahead density and biomass. Eighteen treatments (six defoliation levels, three seasons of defoliation) were applied to 21.5-ft^2 plots on two sites with varying clay content. Plants were clipped in 2004 and 2005. Crested wheatgrass was hand clipped to defoliation levels of 0%, 20%, 40%, 60%, 80%, and 100% in the spring, summer, and fall. Density of crested wheatgrass and medusahead was sampled in June 2005 and 2006, but their biomass was harvested only in 2006. Over the two seasons, site had much more of an impact on medusahead invasion than either defoliation intensity or timing of defoliation. The results support previous suggestions that clayey soils favor medusahead and that perennial grasses with high biomass can resist this invasive species. On the clayey site where medusahead did persist, fall defoliation of crested wheatgrass reduced the density of this invasive species by 50% or more compared to spring defoliation. Given the developmental pattern of medusahead, the goal of any management program should be to maximize resource use by the desirable plant species from April to late July.

INTRODUCTION AND OBJECTIVES

Throughout the Great Basin and the surrounding ecosystems, a major factor affecting rangeland resources, fires, and watershed functioning is invasion by the winter annual grass, medusahead. This annual grass has invaded millions of acres throughout the western United States, and continues to spread at a rapid rate. Within the sagebrush steppe, medusahead aggressively displaces perennial grasses by preempting soil resources, and frequent fires destroy the shrub portion of the plant community. Medusahead-dominated sites have 50 to 80 percent less grazing capacity than the original native plant community. Most ecologists believe that medusahead reduces plant and animal diversity and richness, reduces suitable habitat for wildlife, accelerates erosion, and alters nutrient cycles, hydrologic cycles, and energy flow.

Managers believe this invasive plant species is becoming increasingly common on clay loam and loam soils. However, it might be more competitive and persistent on clay soils even though it has the capacity to encroach in native shrub-steppe plant communities on loam soils. It invades disturbed areas, and in the absence of competition, medusahead demonstrates geometric population growth.

Timing, intensity, and frequency of defoliation affect the competitive interactions between invasive species and perennial grasses, and thus influence the ability of a perennial grass to withstand invasive plant species invasion. An appropriate combination of timing, intensity, and frequency of grazing should allow desired species to remain competitive with invasive species. However, little is known about the effects of defoliation on medusahead establishment in stands of perennial grasses. The objective of this study was to determine the effects of crested wheatgrass defoliation intensity and timing on medusahead density and biomass.

METHODS

This study was conducted from 2004 to 2006 on two sites. Both sites were within the Wyoming big sage/bluebunch wheatgrass community types of eastern Oregon. Site one was located near Venator, Oregon near the Coleman Creek Ranch (lat 43° 33' 49.823" N, long 118° 12' 42.730" W), and site two was located near the south end of Warm Springs Reservoir (lat 43° 26' 15.364" N, long 118° 17' 48.053" W) near Riverside, Oregon. This site, especially with clay soils, is susceptible to invasion by medusahead.

Eighteen treatments (six defoliation levels, three seasons of defoliation) were applied to 21.5-ft^2 plots. Plants were clipped in 2004 and 2005. Crested wheatgrass was hand clipped by weight to defoliation levels of 0%, 20%, 40%, 60%, 80%, and 100% in the spring, summer, and fall. Density was sampled in 2005 and 2006 and on 23 June 2006, the aboveground biomass of crested wheatgrass and medusahead was harvested, dried and weighed.

RESULTS

Biomass Removed

In 2004, the amount of biomass removed was similar at both sites during each season of defoliation (Fig. 1). Except for the summer of 2004 at Coleman Creek, the clipping treatment resulted in a continuous increase in biomass removed, but the amount of biomass removed did not always significantly differ from the adjacent defoliation level. In the summer at Coleman Creek, defoliation levels targeted to receive 60%, 80%, and 100% crested wheatgrass removal received the same clipping intensity. In 2005, across defoliation treatments, Coleman Creek yielded less crested wheatgrass biomass when clipped in the spring or fall than in 2004. At Warm Springs, crested wheatgrass removed was higher in 2005 than in 2004 across all defoliation intensities. However, the amount of biomass removed in the spring of 2004 was the same as that removed in the spring of 2005 at Warm Springs.

Crested Wheatgrass and Medusahead, 2005

In 2005, the only factor that affected crested wheatgrass or medusahead density was site. At Coleman Creek, crested wheatgrass produced 129 tillers per 10.8 ft^2, whereas it produced 228 tillers per 10.8 ft^2 at Warm Springs when averaged across all treatments that year. Medusahead produced 125 plants per 10.8 ft^2 at Coleman Creek and only 20 plants per 10.8 ft^2 at Warm Springs in 2005.

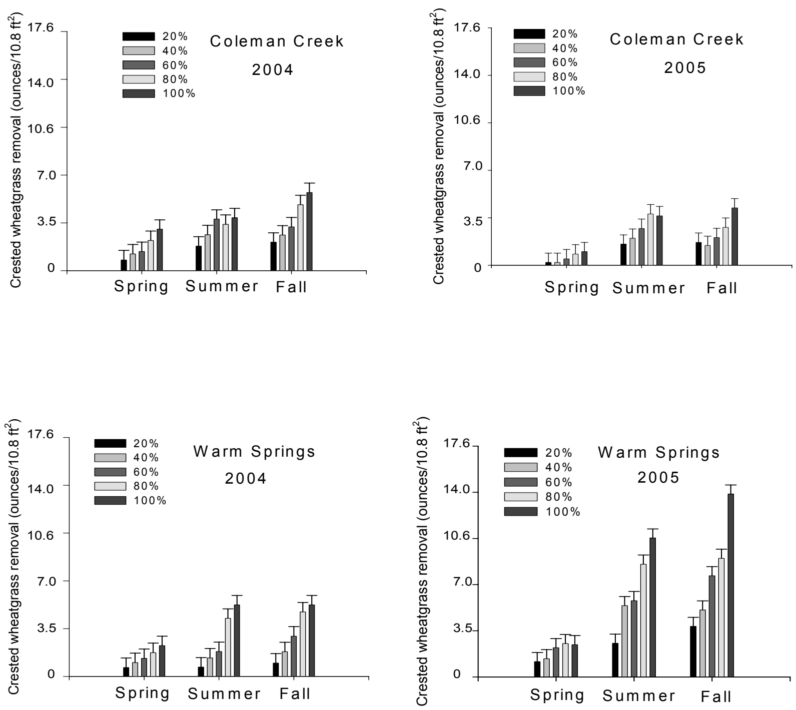


Figure 1. Biomass of crested wheatgrass removed from plots at Coleman Creek and Warm Spring in 2004 and 2005. Error bars ± 1 SE.

Crested Wheatgrass and Medusahead, 2006

Crested wheatgrass density and biomass depended on site in 2006. This grass produced 122 tillers per 10.8 ft^2 at Coleman Creek and 366 tillers per 10.8 ft^2 at Warm Springs when averaged across all treatments. Crested wheatgrass biomass followed a similar pattern that year. It produced 0.82 oz. per 10.8 ft^2 at Coleman Creek and only 0.15 oz. per 10.8 ft^2 at Warm Springs when averaged across all treatments.

Medusahead density and biomass depended on site and interacted with season of defoliation, but not defoliation intensity in 2006. At Coleman Creek, defoliating crested wheatgrass in either the spring or summer yielded about twice the number of medusahead plants than defoliating the bunchgrass in the fall (Fig. 2). By 2006, there were no detectable medusahead plants at Warm Springs.

Clipping crested wheatgrass in the summer yielded the higher medusahead biomass at Coleman Creek, which was about 0.02 oz. per 10.8 ft^2 (Fig. 3). Spring crested wheatgrass defoliation reduced the medusahead yield to about 0.02 oz. per 10.8 ft^2, whereas defoliation in the fall produced about about 0.01 oz. per 10.8 ft^2 of this invasive plant species. Because there were no medusahead plants at Warm Springs, there was no biomass of this invasive species in 2006.

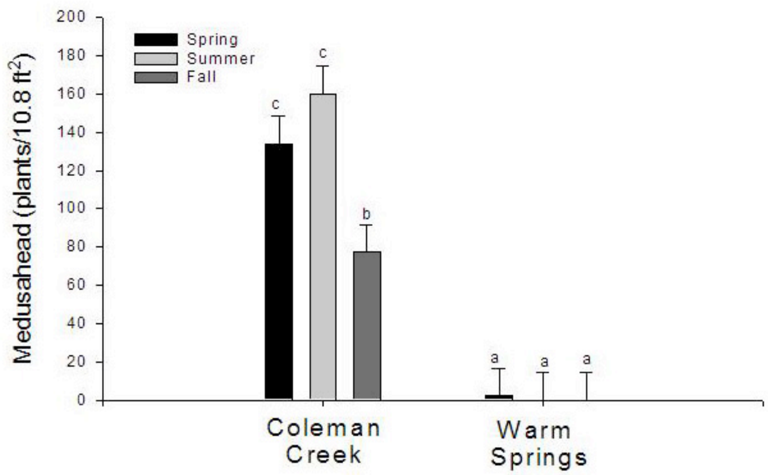


Figure 2. Effect of season of defoliation on medusahead density in 2006. Error bars represent ± 1 SE.

MANAGEMENT IMPLICATIONS

Our study supports the conclusion that on clayey loams and loamy soils, established crested wheatgrass is probably capable of resisting invasion by medusahead if the plants are managed to allow them to fully regain their biomass production from one grazing season to the next. We believe that the response of crested wheatgrass to defoliation, and potentially grazing, and corresponding invasion of medusahead follows a bell-shaped curve on these soil textures. Heavy repeated defoliation in the spring prevents crested wheatgrass from fully recovering its biomass production by the following grazing event and allows invasion. On the other end of the curve, no defoliation allows crested wheatgrass to become old and decadent, and in turn, impedes its ability to rapidly grow and develop a competitive root system in the spring. Our study shows that periodic defoliation of crested wheatgrass is required to maintain enough young, vigorous growth to successfully outcompete medusahead. At one site, defoliating crested wheatgrass in the summer or fall stimulated enough aggressive growth to completely remove all medusahead that had established in the prior year. Without other major disturbances, moderate to heavy grazing intensity applied to crested wheatgrass while alternating the season of use should prevent invasion of medusahead on clay loam soils.

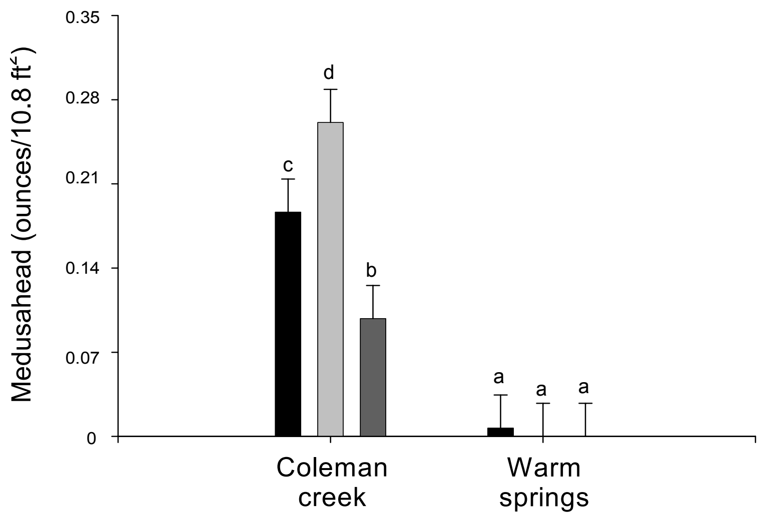


Figure 3. Effect of season of defoliation on medusahead biomass in 2006. Error bars represent ± 1 SE.

REFERENCES

Briske, D. D. and J. H. Richards. 1995. Plant responses to defoliation: a physiological, morphological and demographic evaluation. In: D. J. Bedunah and R. E. Sosebee [EDS]. Wildland plants: physiological ecology and developmental morphology. Denver, CO, USA: Society of Range Management. P. 635-710.

Cook, C. W., L. A. Stodart, and F. E. Kinsinger. 1958. Reponses of crested wheatgrass to various clipping treatments. Ecological Monographs 28:237-272.

Jacobs, J. S. and R. L. Sheley. 1999. Grass defoliation intensity, frequency, and season effects on spotted knapweed invasion. Journal of Range Management 52:626-632.

Western Juniper Ecology and Management

A History of Woodland Dynamics in the Owyhee's: Encroachment, Stand Closure, Understory Dynamics, and Tree Biomass

Rick Miller, Jaime Ratchford, and Dustin Johnson

INTRODUCTION

Piñon and juniper woodlands in the cold desert of the Intermountain West occupy over 44.6 million acres (Miller and Tausch 2001). These woodlands are commonly associated with sagebrush communities forming a mosaic of shrub-steppe and woodland across the region. Numerous studies have documented the recent expansion (since the late 1800's) of these woodlands that has resulted in the replacement of shrub-steppe communities. Recent debate has challenged the degree of expansion in terms of percent of new areas occupied by trees and the increase in total population of piñon and juniper since the late 1800's. Various interest groups have become concerned over the limited scientific evidence documenting the expansion of these conifers at a broad scale (in other words, landscapes or across entire woodlands) in the Intermountain Region. The fear of many groups is historic woodlands that occupied landscapes prior to Eurasian settlement in the late 1800's are being burned, cut, and chained in the name of restoration.

To evaluate the magnitude of expansion on a regional level we evaluated six woodlands from their lower to upper elevational boundaries in four different ecological provinces (Miller et al. 2008). In this report we summarize our findings of woodland expansion in the Owyhee Mountains and discuss our preliminary findings from an ongoing study documenting changes in plant composition, structure, biomass, and fuel loads with increasing tree dominance. Specific questions addressed in this report are:

1. What was the density and spatial extent of trees prior to 1850?
2. What was the chronological sequence of tree establishment and rates of expansion into shrub steppe communities during the past 150 years?
3. What is the current successional state of woodland development (Phase I – early, II – mid, III – late successional)?
4. How do plant cover, structure, and biomass change in relation to woodland succession (Phase I, II, and III)?

STUDY AREA

The study areas are located on Juniper and South Mountain in the Owyhee Mountain Range in Owyhee County, Idaho and are considered part of the Humboldt Ecological Province (Fig. 1). The geomorphology of this area is characterized as an uplifted region with doming and fault blocking common. The Owyhee Mountains are predominantly comprised of granite; however, most of the uplands are overlain by rhyolites and welded tuffs with silicic volcanic flows, ash deposits, and wind-blown loess. Topographic characteristics of this area include mountains dissected by deep canyons, rocky tablelands, and rolling plains ranging in elevation from 3,936 and 7,790 ft.

Climate across the Owyhee Mountains is characteristic of the northern Great Basin in that it is cool and semiarid. Mean annual precipitation within the juniper belts varies between 12 inches at lower elevations increasing to 16 inches at higher elevations. The majority of the annual precipitation is received as snow in November, December, and January and as rain March through June. Average temperatures vary from 20.2°F in January to 94.1°F in July. The growing season varies from 90 to 120 days. Soils range from shallow rock outcrops to moderately deep gravelly, sandy, or silt loams. Predominant soil taxa are Aridisols, Entisols, Alfisols, Inceptisols, and Mollisols that occur in combination with mesic and frigid soil temperature regimes and xeric and aridic soil moisture regimes. The National Resources Conservation Service has described the area's potential natural vegetation as sagebrush-

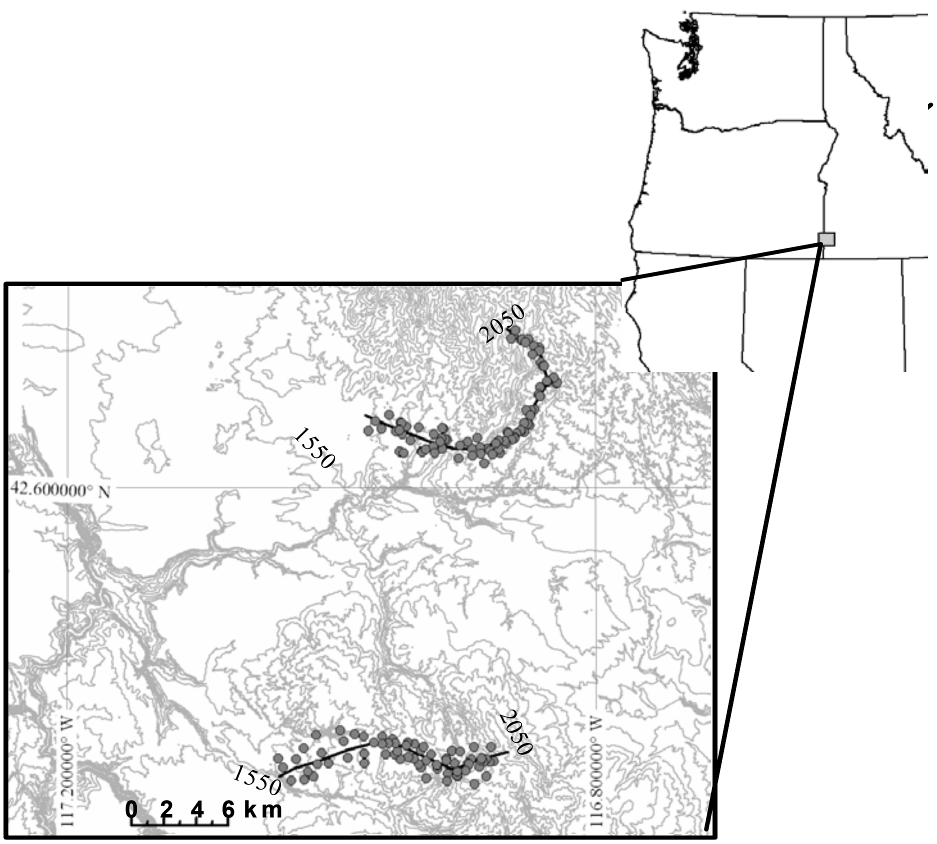


Figure 1. Map of the study locations on South Mountain (north transect) and Juniper Mountain (south transect) in southwest Idaho. Three circular plots were placed approximately every 0.3 miles along 3 parallel transects spaced 0.3 miles apart; plot locations were adjusted to fit within a uniform stand at least 1.25 acres in size with uniform characteristics (e.g. aspect, topography, soil, and vegetation).

grassland. Predominant vegetation in the area is western juniper, mountain big and low sagebrush, Idaho fescue, bluebunch wheatgrass, western needlegrass, Thurber's needlegrass, and Sandberg bluegrass.

METHODS

To gain a landscape scale perspective of both the spatial expansion and increasing density of juniper, we established two transects each approximately 10 miles long across two woodlands on Juniper and South Mountain. Each transect was located along an elevational gradient that extended from the lower to upper boundaries of each woodland. Across the two transects we sampled tree age structure and density. Expansion of post-settlement woodlands was determined by aging (coring and counting the rings) of the three largest trees with post-settlement morphological characteristics. This enabled us to estimate when the first post-settlement trees established on the plot. Tree density was measured by counting live and dead trees in 158, 0.2- to 0.7-acre plots (plot size varied with tree density). Stand density measurements included presence, absence, and density of trees establishing prior to and after 1850, standing dead, stumps, and logs. A complete age structure of trees was measured on Juniper Mountain by aging all trees within the plots. In an ongoing study, near Juniper Mountain we are measuring overstory and understory structure (cover and density) and biomass in 45, 0.25- acre plots.

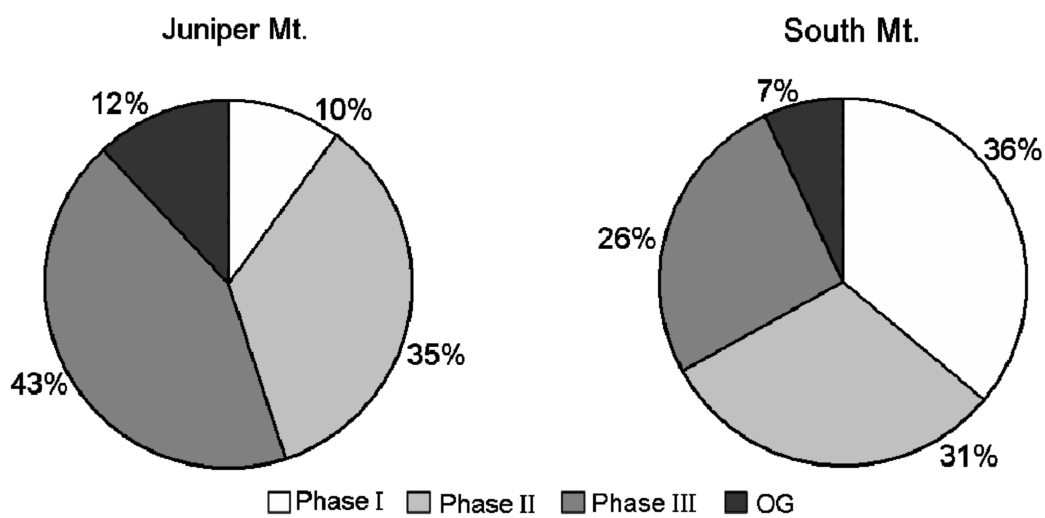


Figure 2. Proportion of woodlands in four successional states; Phase I = trees present but shrubs and grasses dominate the site, Phase II = trees co-dominate the site with shrub and grasses, Phase III = trees dominate the site and shrubs and grasses have declined, and OG = stands with ≥ 75% of the trees older than 150 years.

RESULTS AND DISCUSSION

Presettlement Western Juniper Stands

Prior to 1850, 12% and 7 % of the landscape in Juniper Mountain and South Mountain, respectively, were occupied by juniper woodland (Fig. 2). The remaining 88 to 93% were dominated by shrub-steppe and grasslands with an intermingling of scattered western juniper. Within these shrub-steppe and grassland communities 48 and 67% on Juniper and South Mountain, respectively were occupied by a low density of scattered juniper trees prior to 1850 (Fig. 3a). Of the current population of trees greater than 3 ft tall, 5 and 10% established prior to 1850 (Fig. 3b). Both density and frequency of occurrence of western juniper prior to 1850 were greater across the two Idaho woodlands compared to woodlands measured in southeastern Oregon where pre-1850 trees occurred in less than 30% of the stands measure and 2% or less of the current population of trees (Johnson and Miller 2007).

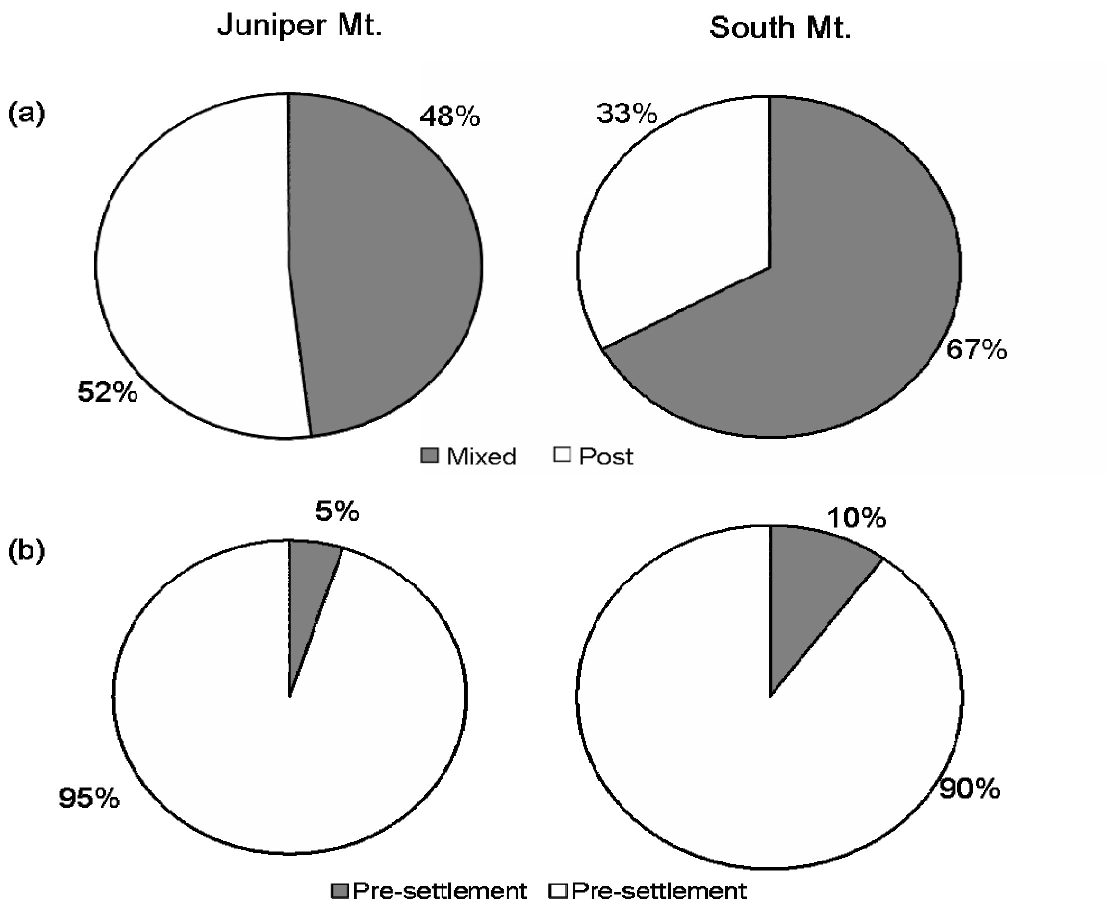


Figure 3. The proportion of (a) mixed age (contained at least on tree ≥ 150 years old in a 0.7 acre plot) and post settlement age stands (trees < 150 years); (b) percent of the total tree population ≥ 3 ft tall that were presettlement (≥ 150 years old) and post-settlement (<50 years old).

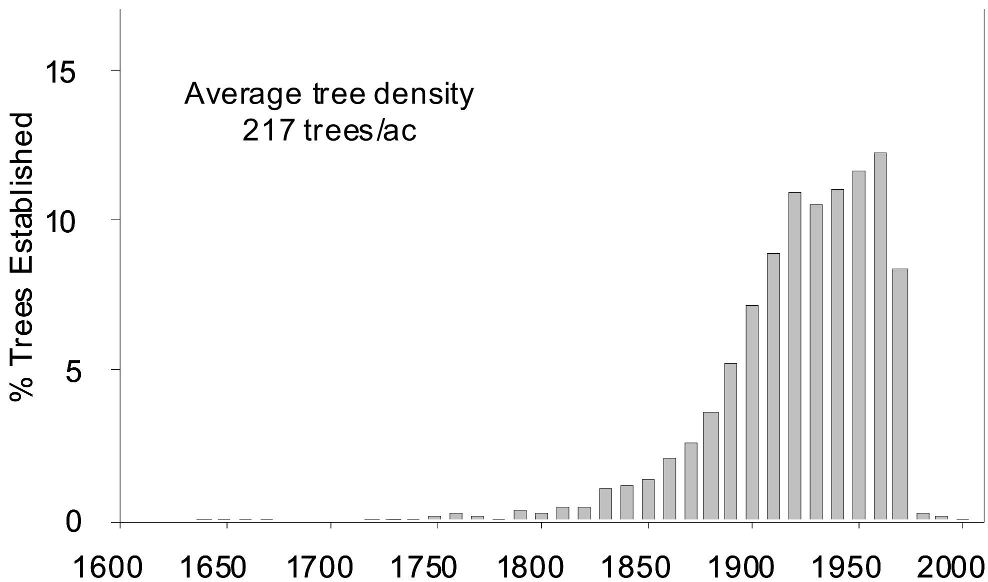


Figure 4. Current mean juniper density and decadal establishment on Juniper Mountain, Idaho.

Pattern of Expansion and Establishment

On Juniper Mountain where we did the intensive age structure sampling, current density of trees is 217/acre. Since 1850, tree densities have increased more than 10 fold. There was a slight increase in the rate of establishment in the mid 1800s, which then increased rapidly during the late 1800s and early 1900s (Fig. 4). The sudden decrease in tree establishment in the past three decades is largely a result of the large proportion of stands that are closed or approaching closure (late Phase II and Phase III). Competition among overstory trees reduces seed crops and tree seedling establishment. The rate of tree expansion into treeless shrub-steppe communities peaked between 1880 and 1930 (Fig. 5). The decline is a result of a shrinking proportion of the landscape without trees. Tree establishment (number of trees establishing/year) rates increased with elevation and a shift from southerly to northerly aspects (Fig. 6). Currently over half of the stands measured on both Juniper and South Mountains are in Phase II and III with a third or less in Phase I (Fig. 2).

Overstory-Understory Relationships

Many of the closed stands today shifted from Phase II to Phase III in the mid 1950s. This is based on a sharp decrease in the relative annual growth rates in the 1950s, which continue to remain low compared tree growth rates in Phase I stands (Fig. 7). The decline is probably caused by intra-specific competition among trees, a result of limited soil resources. As soil nutrients and water become increasingly limited, the abundance of understory vegetation declines. The relationship between understory (shrubs and herbaceous plants) and overstory cover (trees) shown in Figure 8 is for a mountain big sagebrush/Idaho fescue plant association near Juniper Mountain. The maximum juniper cover measured across 31, 0.25 ac

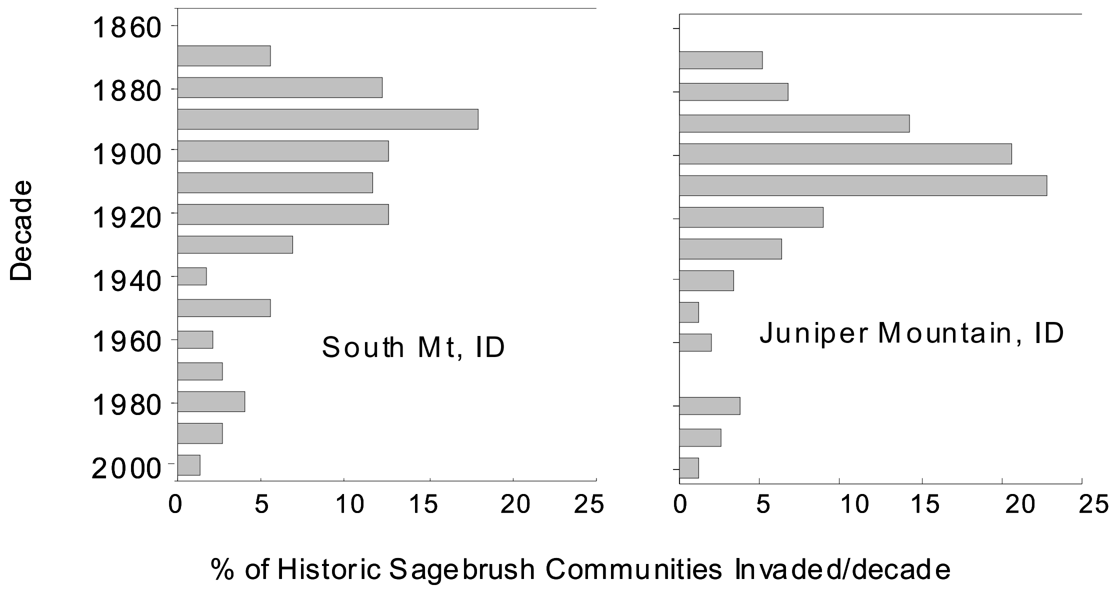


Figure 5. The proportion of decadal encroachment of juniper between 1860 and 2000 into treeless (no evidence of presettlement trees) sagebrush-steppe communities.

plots approached 70% on Juniper Mountain. Similar values have been reported by Miller et al. (2000) for this plant association in southeastern Oregon and northeastern California. Variation of understory cover within phases of woodland development is partially a result of different soil characteristics, especially depth to a restrictive layer. However, the graph illustrates that a shift from Phase I to II occurs at about ¼ of maximum potential juniper cover (approximately 15% tree cover). The shift from Phase II to III occurs at about one-half of maximum potential cover (approximately 30% tree cover). Tree biomass in Phase I was below 9,000 lbs/acre and increasing to over 30,000 lbs/acre in phase III (Fig. 8).

ACKNOWLEDGEMENTS

Both past and ongoing studies are funded and supported by the Joint Fire Science Program and Eastern Oregon Agricultural Research Center.

REFERENCES

Johnson, D.D., and R.F. Miller. 2006. Structure and development of expanding western juniper woodlands as influenced by two topographic variables. Forest Ecology and Management 229:7-15.

Johnson, D.D., and R.F. Miller. 2008. Old-growth juniper: distribution, abundance, and influence on post-settlement expansion. Journal of Range Ecology and Management 61:82-92.

Miller, R.F., and R.J. Tausch. 2001. The role of fire in pinyon and juniper woodlands: a descriptive analysis. Tall Timbers. Pages 15-30, *in* K. Galley and T. Wilson (eds.), Fire Conference 2000: The First National Congress on Fire, Ecology, Prevention, and Management. Invasive Species Workshop: The Role of Fire in the Control and Spread of Invasive Species. Tall Timbers Research Station, Tallahassee, Florida.

Miller, R.F., J.D. Bates, T.J. Svejcar, F.B. Pierson, and L.E. Eddleman. 2005. Biology, ecology, and management of western juniper. Oregon State University Agricultural Experiment Station Technical Bulletin 152. Corvallis, Oregon.

Miller, R.F., R.J. Tausch, D. Macarthur, D.D. Johnson, and S.C. Sanderson. 2008. Development of post settlement piñon-juniper woodlands in the Intermountain West: a regional perspective. USDA Forest Service Research Paper Report RMRS-RP-69.

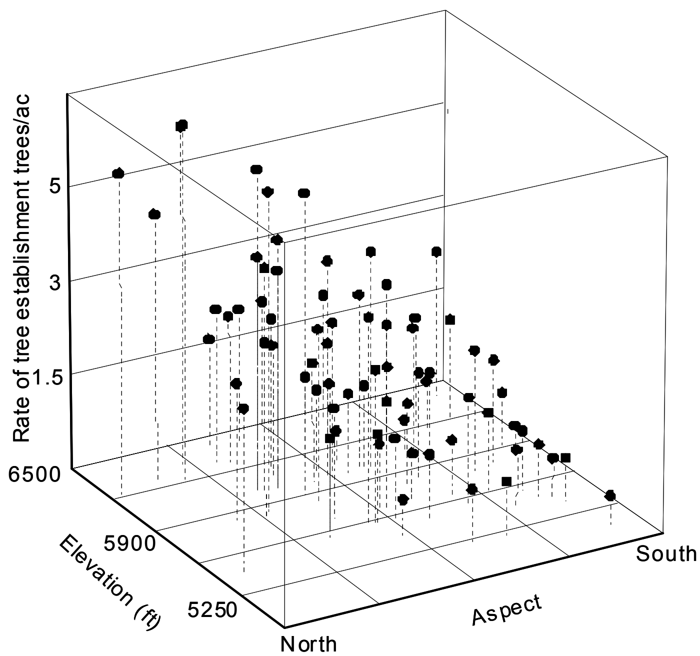


Figure 6. Relationship of tree establishment rates (trees/acre/year) with elevation and site exposure in stands associated with mountain big sagebrush. Site exposure shifts from a northerly to southerly aspect from left to right.

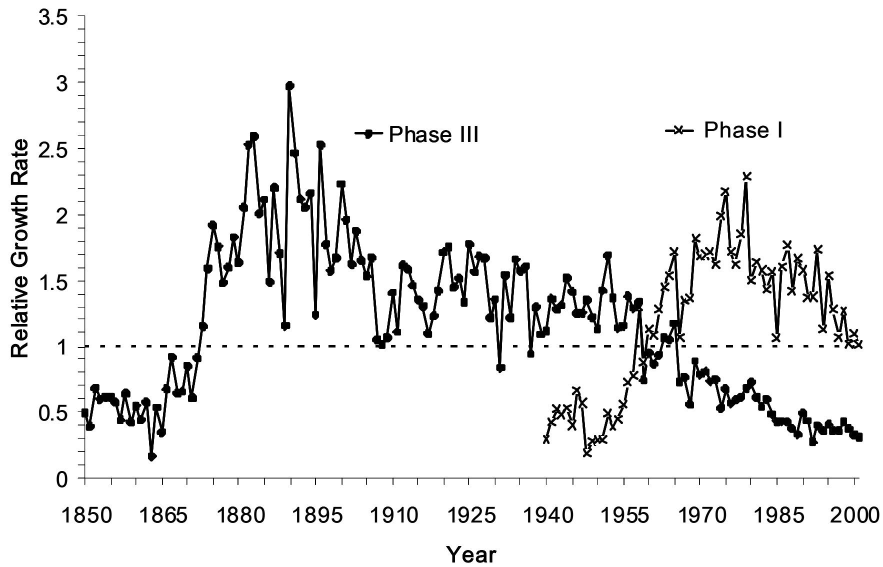


Figure 7. Relative growth rates based on tree ring widths for Phase I and III. Relative growth is typically slow during the first 15-20 years of tree growth. The number 1 is the relative mean ring width for a composite of trees in Phase III. The y-axis is the relative growth rate (or magnitude) of growth compared to the mean. To compare growth rates between Phase I and III, the relative growth rate for trees currently in Phase I was based on the mean growth rate for the Phase III trees.

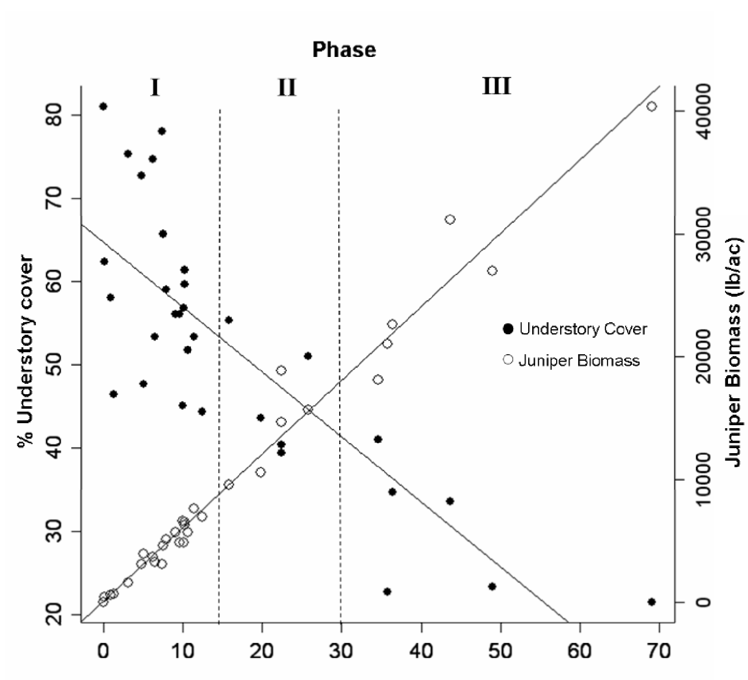


Figure 8. Relationship between overstory juniper cover, total tree biomass, and total understory (shrubs and herbaceous plants) cover. Phase I = trees present but shrubs and grasses dominate the site, Phase II = trees co-dominate the site with shrubs and grasses, Phase III = trees dominate the site and shrubs and grasses have declined.

Juniper Cutting and Prescribed Fire Combinations; South Mountain, Idaho

Jon Bates, Kirk Davies, Roger Sheley, and Rob Sharp

INTRODUCTION

Control of western juniper by burning or cutting has been successfully used to reestablish shrub/understory plant communities in the northern Great Basin. Tree cutting, using chainsaws, has generally been applied to areas that have fully developed woodlands (Phase III) and no longer possess the understory fuels to successfully carry a fire. Woodlands that lack adequate fuels are those in mid- to late successional stages where juniper competition has eliminated the shrubs and reduced understory production. In many juniper control projects only a portion of the trees need to be cut to increase surface fuels so that prescribed fire can remove remaining live trees. Reducing the number of trees cut could lower costs and permit larger acreages to be treated.

An objective of the research was to assess what level of preparatory cutting is required to eliminate remaining juniper trees by prescribed fire in the fall. The study also evaluated the effect of the treatments on post-fire vegetation dynamics and effects of seeding for site recovery.

METHODS

Study sites were set up within the Juniper and Cabin Creek drainages on South Mountain, Idaho (about 6000 ft. elevation). Two plant community types were selected for treatment; western snowberry-mountain sagebrush/Idaho fescue-western needlegrass (SNOWBERRY) and mountain big sagebrush/letterman's needlegrass (SAGE). Sites were dominated by post-settlement western juniper woodlands (all Phase III woodlands). Phase I woodlands contain trees but shrubs and herbs are the dominant vegetation. In Phase II, trees are co-dominant with shrubs and herbs and all three vegetation layers influence ecological processes. Phase III woodlands are when juniper is the dominant vegetation and the primary plant layer influencing ecological processes.

Juniper cover ranged between 40-70% and tree densities ranged from 100-300 trees per acre. Preparatory tree manipulations were chainsaw cutting 25%, 50%, and 75% of the post-settlement trees in October 2002 (treatments were: SAGE 25, SAGE 50, SAGE 75; SNOWBERRY 25, SNOWBERRY 50, SNOWBERRY 75). Each treatment plot was 1.5 acres in size and was replicated 5 times (40 plots total). Total area used per community type was 30 acres (60 acres total). Cut trees were allowed to dry for one year prior to fire application. Prescribed fires (strip head fire) were applied on October 21-22, 2003. Uncut/unburned woodlands (CONTROL) were located adjacent to treated areas.

Sampling included measurement of cover and density of juniper, shrubs, and herbaceous species. Herbaceous species were also measured for biomass (2006) and diversity. One year of pre-treatment (2002) and three years of post-treatment response (2004-2006) data were collected.

Seeding trials were conducted on a small portion of the study area. Six native species were seeded in monoculture at four rates and seeded in a mixture of all six species at four rates. Non-seeded controls were established to compare untreated response with seeded plots. Native grasses were bluebunch wheatgrass, Idaho fescue, Sherman big bluegrass. Forbs were western yarrow, arrowleaf balsamroot, and wild blue flax. Seeding rates were 15.6, 20.4, 25.4, or 31.1 lb/acre of pure live seed.

RESULTS

Juniper Removal

The partial cut and burn treatments were all successful at removing remaining live western juniper trees. On the SNOWBERRY type the fire killed 95-99% of remaining live trees and juniper cover was reduced by 99%. On the SAGE type fires killed 85-100% of the remaining live trees and juniper cover was reduced by 90-94%.

Understory Dynamics

Understory response did not differ among the various cutting levels and prescribed fire applications within each plant community type. In both communities, perennial bunchgrass densities were moderately to severely reduced the first year after burning. However, by the third growing season (2006) after fire, perennial grass densities were greater in the treated sites than the CONTROL as a result of large numbers of grass seedlings.

Herbaceous biomass was greater in treated sites than the CONTROL in 2006 (Fig 1). On the SNOWBERRY type annual forb and total biomass were greater than the CONTROL (Fig. 1A). On the SAGE type perennial bunchgrass, perennial forbs, annual forbs, and total biomass were greater than the CONTROL (Fig. 1B).

In the SAGE community type, cover and density of perennial forbs, annual forbs, and total herbaceous were greater in the burned treatments than the CONTROL (Fig 2). Forb diversity increased by 30%. Weeds slowly increased but were not expected to dominate either plant community type. Cheatgrass increased on the SAGE type; however, because of increased establishment of perennial grass seedlings cheatgrass is not expected to become dominant. Bare ground did not increase as a result of the fire; however, juniper litter was reduced after burning (Fig. 3).

In the SNOWBERRY community type, cover and density of annual forbs and total herbaceous were greater in the burned treatments than the CONTROL (Fig. 4). Forb diversity increased by 10-22%. Bare ground was greater in the treated plots than the CONTROL (Fig. 5). Juniper litter was reduced after burning. Herbaceous litter increased in the treated areas and was greater than the CONTROL in 2006.

Shrub cover and density has not differed among treatments for either community type. Sagebrush and bitterbrush remained present in the SAGE type but densities were reduced by burning

.

Seeding

Bluebunch wheatgrass, Idaho fescue, Sherman big bluegrass, western yarrow, and wild blue flax all established successfully. Arrowleaf balsamroot did not establish successfully. The highest biomass production was combination seeding at 20.4 lb/acre on the SAGE site and 31.1 lb/acre on the SNOWBERRY site. Seeding a combination of species resulted in a moderate to high density of plants and optimized plant diversity and richness over seeding monocultures.

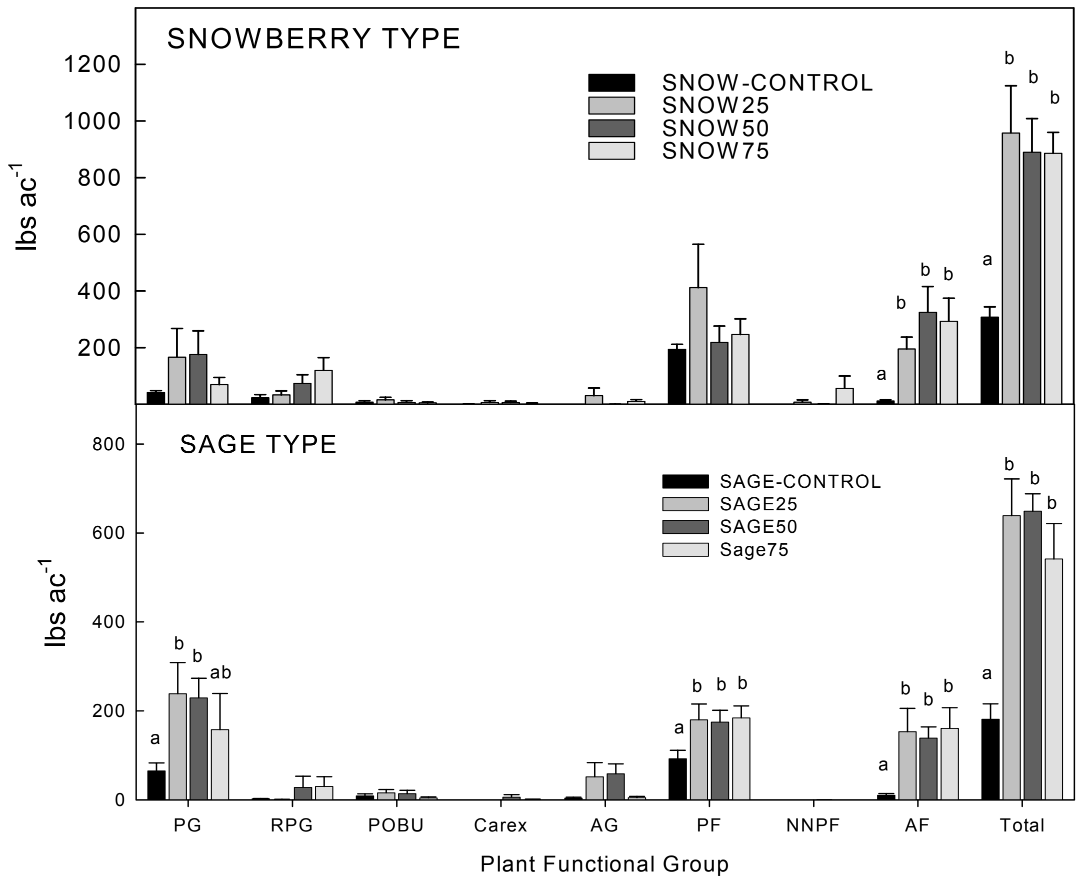


Figure 1. Biomass (lb ac^{-1}) of the functional group biomass in 2006. Data are in means + one standard error. Significant differences among treatments for functional groups are indicated by different lower case letters. Functional groups are; Perennial Bunchgrass (PG); Rhizomatous grasses (RPG); Bulbous bluegrass (Pobu); Carex spp.; Annual Grasses (AG); Native Perennial Forb (PF); Non-Native Perennial Forb (NPF); Annual Forb (AF); and Total Herbaceous.

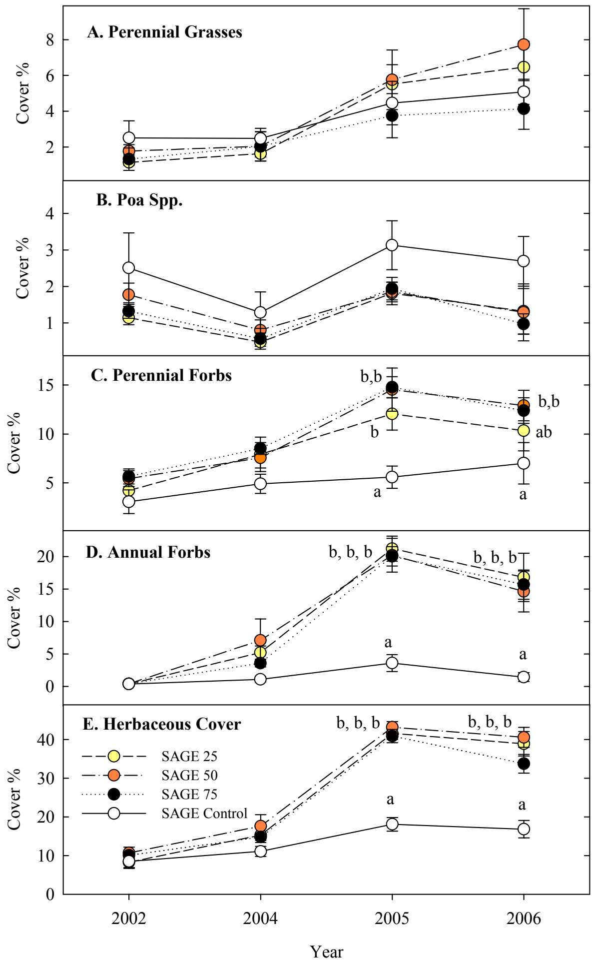


Figure 2. SAGE TYPE: Functional group cover (%) for the treatments in 2002-2006. for: (A) Perennial Grasses; (B) Bluegrass spp. (Bulbous and Sandberg's); (C) Perennial Forbs; (D) Annual Forbs; and (E) Total Herbaceous. Data are in means $\pm$ one standard error. Cut and burn treatments did not differ but all are greater than the Control for Perennial Forbs, Annual Forbs, and Total Herbaceous.

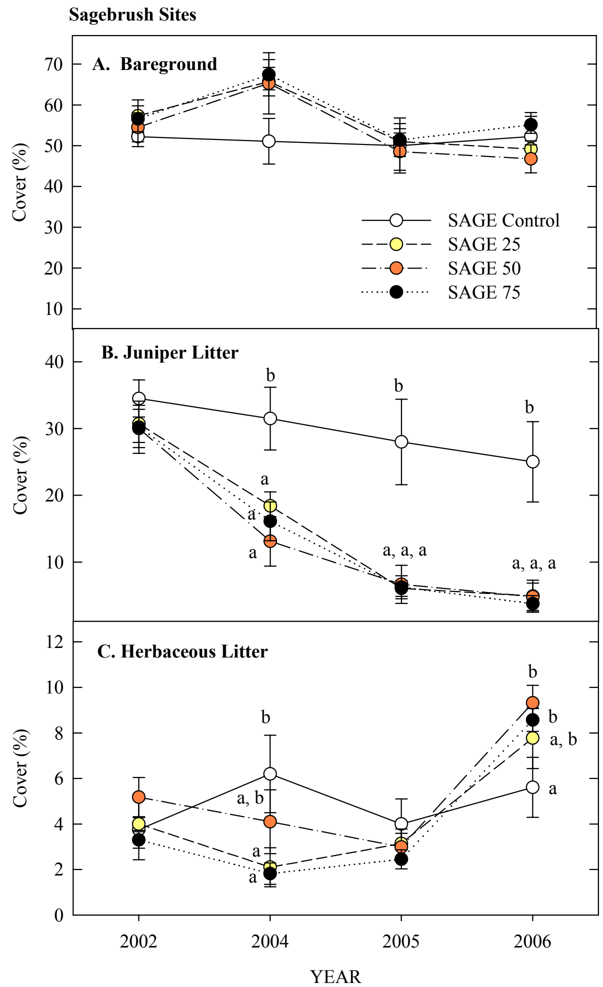


Figure 3. Sagebrush Sites: Ground cover (%) for the treatments in 2002-2006. for: (A) Bareground; (B) Juniper Litter; and (C) Herb. Litter. Data are in means $\pm$ one standard error. Juniper litter was reduced by burning.

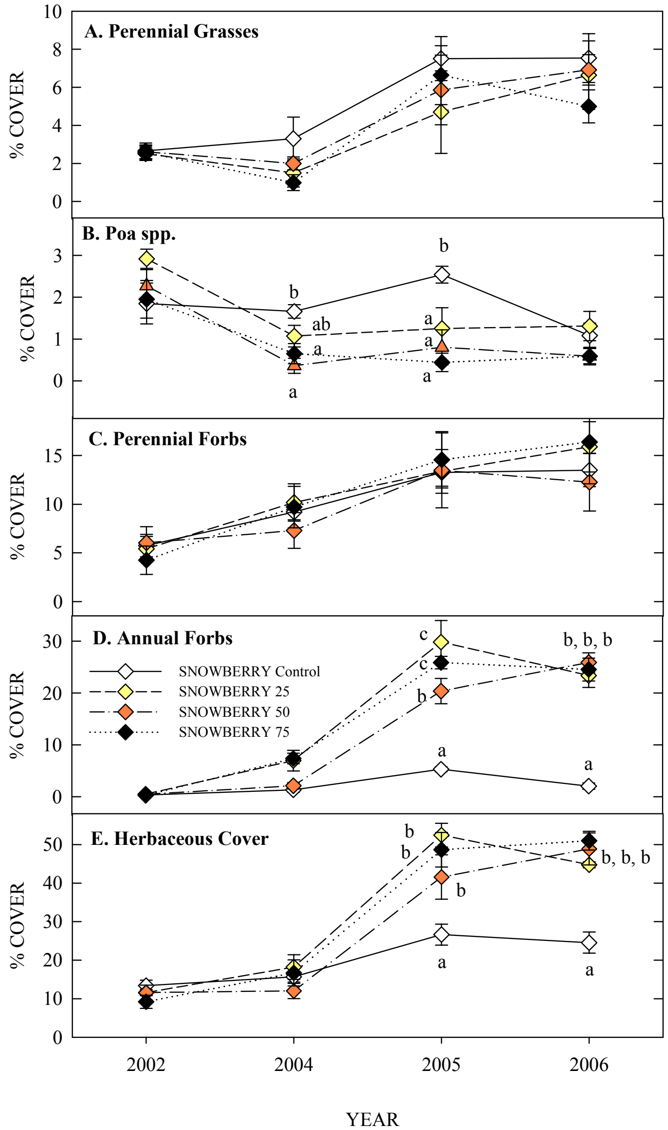


Figure 4. Snowberry Sites: Functional group cover (%) for the treatments in 2002-2006. for: (A) Perennial Grasses; (B) Bluegrass spp. (Bulbous and Sandberg's); (C) Perennial Forbs; (D) Annual Forbs; and (E) Total Herbaceous. Data are in means ± one standard error. Cut and burn treatments did not differ but they all are greater than the Control for Annual Forbs and Total Herbaceous.

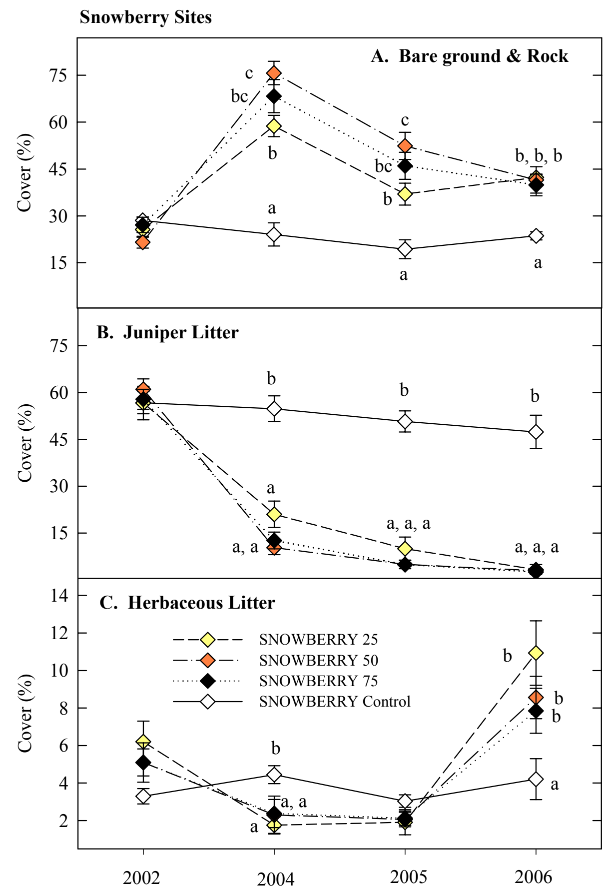


Fig 5. Snowberry Sites: Ground cover (%) for the treatments in 2002-2006. for: (A) Bare ground; (B) Juniper Litter; and (C) Herb. Litter. Data are in means ± one standard error. Bare ground was increased by the fire primarily because of the large reduction in juniper litter. Herbaceous litter increased in 2006 in the treated areas.

DISCUSSION and CONCLUSIONS

Juniper Control

The results indicated that cutting only 25% of mature trees in communities dominated by western juniper (Phase III, juniper cover 40-70%) was sufficient to remove the majority of remaining live trees in stands with fall prescribed fire. Cutting more than 25% of the trees was excessive when broadcast burning was applied with weather conditions typically encountered with fall prescribed fire. Burning was equally effective on slopes (SNOWBERRY and SAGE type 10-50% slopes) and on flat ground (SNOWBERRY type). Crowning or canopy fires were maintained in some adjacent uncut woodlands with little understory. Cutting levels could probably be reduced in the areas with greater than 60% juniper cover (e.g. SNOWBERRY type, about 300 stems per acre) and still achieve sufficient kill on remaining live western juniper. Cutting 15-25% (about 50-75 stems cut per acre) of the trees or strip cutting should be considered when fall burning thicker stands of western juniper. For treating western juniper on a large scale cutting-prescribed fire combinations are a useful method of tree control that should reduce treatment costs.

Old Juniper Trees

A portion of the old western juniper trees growing on rocky outcrops above the treatment areas were killed by the fire. These trees had survived historical (pre-settlement) fire events. Cut trees when burned are obviously providing the necessary heat and flame lengths to kill the older trees. Options to increase survival of the older trees would be to physically remove cut western juniper fuels adjacent to old trees or not cutting post-settlement trees growing near old trees until after fall fire is applied to surrounding areas. The invasive trees could then be cut and winter burned.

Understory Dynamics

The results present early successional dynamics after cut-and-burn treatments. These treatments produced severe impacts to the understory and it is clear that understory recovery will take longer than 3 years. On the SAGE type, hot spots such as around tree boles and under cut trees are expected to permit cheatgrass to dominant these locations for several years. However, the potential for plant community recovery is high because of the sites elevation and precipitation zone, and present herbaceous composition. As this point it appears that both the SAGE and SNOWBERRY types will recover with primarily native perennial vegetation.

When prescribing these cut-and-fall burn treatments in Phase III woodlands it can be initially be expected to stimulate perennial and/or annual forbs; perennial bunchgrasses will be moderately to severely reduced. If cheatgrass or medusahead is present there is potential for these species to take over a site, thus, management should be cautious when applying these treatments.

Seeding

To speed site recovery after fire seeding should be considered and livestock grazing should be carefully managed. On both community types seeding trials were highly successful at speeding perennial bunchgrass response and were observed to reduce erosion.

Other Management Considerations

For management purposes it is most important to increase perennial grass cover/densities as this functional group has the most value at reducing erosion and minimizing weed invasion in Great Basin plant communities. Until ground cover is reestablished (5-8 years) burned areas can experience significant runoff with moderate to heavy rain showers.

Because of the high mortality of perennial bunchgrasses in response to fire, grazing rest or deferrals will likely be necessary after fire treatments. Grazing management of burned areas should be flexible; with the overall objective of increasing perennial grass cover/densities to a minimum of 4-6 plants/yard^2. Both sites are capable of supporting higher perennial grass densities (10-20 plants/yard^2). With or without seeding, burned areas should probably be rested the first and possibly second year after fire to maximize bunchgrass seed crop and establishment. Deferring grazing until after seed shatter the second through fourth growing season would increase grass seed production and establishment. Based on the results here and in our other studies it will take longer than 2 years, possibly as many as 8 years, for bunch grasses to fully recover. Early season grazing (prior to grass boot stage) or deferring grazing until after seed shatter should probably be considered in later years after fire (years 3-8).

Winter burning of cut trees should be considered to limit mortality of herbaceous perennials and speed recovery (Bates et al. 2007). Cutting levels would have to greater (60-80%) to kill enough trees to elicit an herbaceous/shrub response. Burning should be done when soil and ground litters are at field capacity and frozen (Nov.-March).

ACKNOWLEDGEMENTS

Many thanks are extended to Bill and Tim Lowry (Jordan Valley, Oregon) and Idaho Department of State Lands for providing land for conducting the study. Mike and Jeanie Stanford permitted access through their property and provided logistical help. The success of the project owes much to the many student summer range technicians who assisted in the collection of field data and to ARS range technicians assisting in the fire applications.

REFERENCES

Bates, J., R. Miller, T. Svejcar, K. Davies, F. Pierson, S. Hardegree. 2007. Western Juniper Control Studies: EOARC Research Progress Report. Eastern Oregon Agricultural Research Center, Station Report, Burns. 57 p.

Hydrologic Response to Western Juniper Control

Tim L. Deboodt, M.P. Fisher, J.C. Buckhouse, and John Swanson

SUMMARY

Western juniper (*Juniperus occidentalis)* has been associated with increased soil loss and reduced infiltration resulting in the loss of native herbaceous plant communities and the bird and animal species that rely on them. USDA Forest Service inventory analysis indicates that since 1934 western juniper's dominance on eastern Oregon's rangelands has grown from 1.5 million to over 6 million acres. In 1993, a paired watershed study was initiated in the Camp Creek drainage, a tributary of the Crooked River of central Oregon, to evaluate the impacts of cutting western juniper on the hydrologic function of a site. The study involved a paired watershed approach using watersheds of approximately 240 acres each to evaluate changes in a system's water budget following the reduction of western juniper. Water budget is measured in terms of inputs (precipitation) and outputs (soil moisture, runoff, groundwater recharge and evapotranspiration). Watershed impacts include changes in water budget as well as vegetation composition and cover changes and altered erosion rates. Previous monitoring studies have been limited in their scope to water quality impacts (soil erosion) and infiltration rates and vegetative responses following juniper control.

In 2005, following 12 years of pretreatment monitoring in two watersheds (Mays and Jensen) all post post-settlement aged juniper (< 140 years of age) were cut from the treatment watershed (Mays) (Fig. 1) Hydrologic responses including changes in depth to ground water, spring flow, channel flow, and soil moisture were assessed. Vegetative cover changes were measured. Analysis indicated that juniper reduction increased late season spring flow, increased days of recorded ground water and increased the relative availability of soil moisture at the deeper soil depths. Ephemeral channel flow and channel morphology did not show a predictable trend following 2 years of post-treatment measurements. Precipitation received from October through May accounts for 70 percent of the annual precipitation and directly impacts actual water yields.

INTRODUCTION AND OBJECTIVES

Western juniper's dominance in eastern Oregon has increased 5 fold since 1934 (420,000 acres in 1934 and 2,200,000 acres in 1999). Based on water use models for individual trees, the U.S. Forest Service estimates that mature western juniper tree densities, ranging from 9 to 35 trees per acre, are capable of utilizing all of the available soil moisture on a given site. Research has shown that soil loss from sites with higher than the natural variation of western juniper cover is an order of magnitude greater than similar sites that are still within their natural range of variation.

Water quantity and timing are the primary factors being monitored with this project. The project involves the use of a paired watershed study. The project consisted of the treatment (cutting juniper) of one of the paired watersheds totaling approximately 250 acres with the other watershed serving as the untreated control. The Prineville Bureau of Land

Management (BLM) District cut approximately 200 acres of western juniper in the Mays watershed. The cutting was started in October, 2005 and was completed in April, 2006.

The paired watershed project is located approximately 60 miles southeast of Prineville, Oregon. In 1993, two watersheds (Mays and Jensen) were identified in the Camp Creek drainage. Each watershed is located at the headwaters of each drainage.

The elevation of the project area ranges from 4,500 - 5000 feet with an average annual precipitation of 13 inches. The historic vegetation type was mountain big sagebrush / Idaho fescue. The site is currently dominated by western juniper with a sparse understory of shallow rooted perennial grasses and forbs. Since 1994, the two watersheds have been monitored for similarities and differences.

Project Objectives

- Evaluate hydrologic changes following the cutting of post-European aged juniper (trees established since mid-1800's).
- Evaluate changes in hill-slope erosion and channel morphology following the cutting of post-settlement juniper.
- Evaluate changes in plant community composition following the cutting of post-settlement juniper.

The majority of the two watersheds are comprised of public land, administered by the Prineville District, BLM (75% Mays, 86% Jensen). The remaining portions of each watershed are owned by the Hatfield High Desert Ranch. The BLM, in cooperation with Crook County Soil and Water Conservation District (SWCD), the permittee (Hatfield's), and Oregon State University ()SU) Department of Rangeland Ecology and Management, identified the paired watersheds as an area of interest because of the opportunities the study provided to monitor changes in water yields as a result of juniper control.

METHODS

Establishment of the study began in 1994. Each watershed was delineated by the location of a continuous recording flume placed in the channel at the lowest point of each watershed. Flow was measured and recorded with the aid of a data logger. Precipitation inputs were first measured with the use of Belfort Universal Rain Gauge and in 2004, a weather station was added to each watershed to record air temperature, precipitation, wind speed and direction, solar radiation, leaf wetness, relative humidity and snow accumulation.

Permanent vegetative transects were established in each watershed and located by aspect. In each watershed, 8 – 100 ft. transects were located, 2 per aspect and data were collected in 1995, 2003, 2005, and 2007. Basal cover of grasses and canopy cover of forbs, shrubs, and trees were recorded.

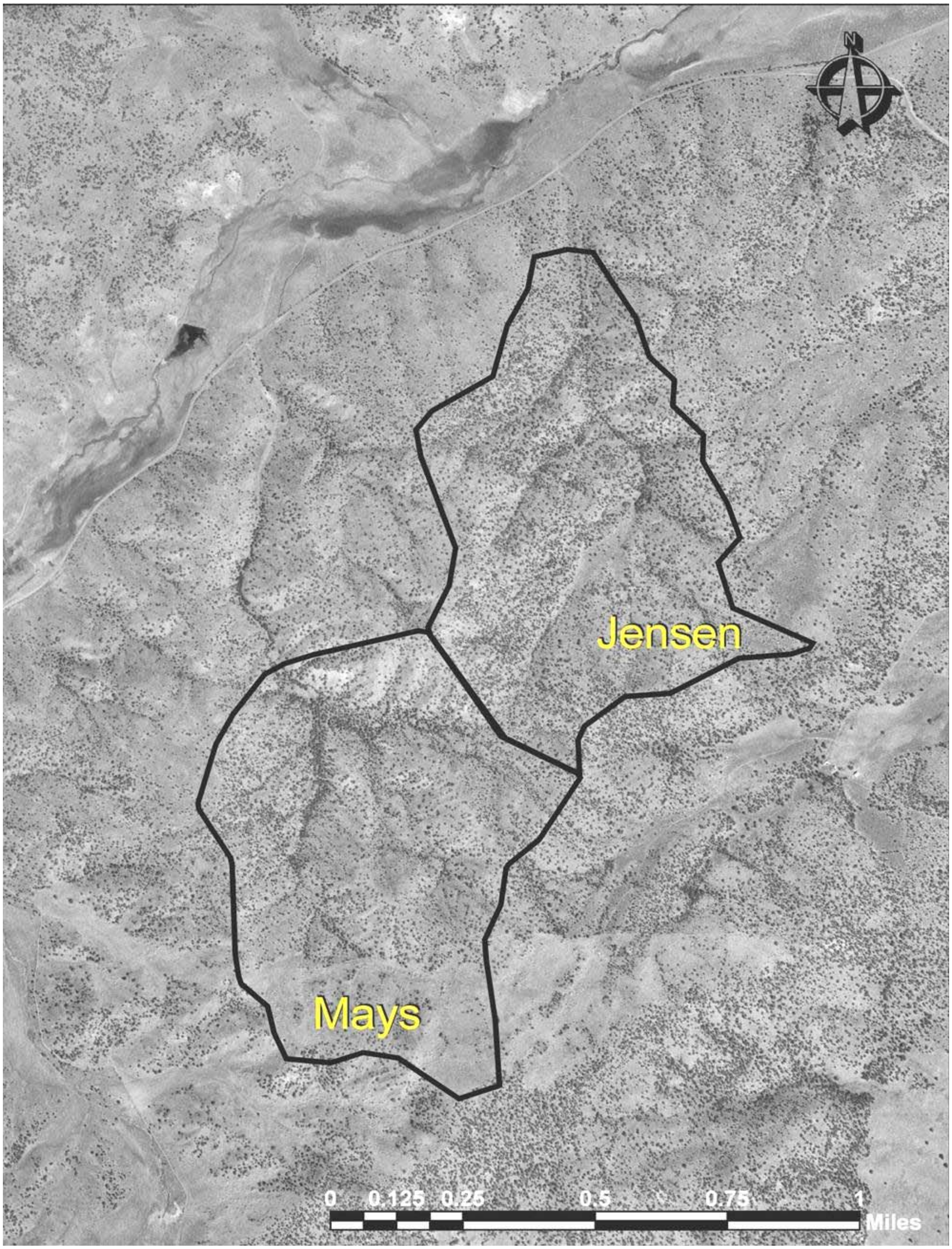


Figure 1. Aerial photograph of project area 60 miles southeast of Prineville, Oregon

Permanent channel cross-sections and hill-slope erosion plots were located in each watershed. Twenty-five cross-sections and twelve sets of hill-slope transects have been measured once or twice a year.

In 2004, additional monitoring was added to the watersheds (Fig. 2). Within each watershed, a spring was improved and flow measured. Six peizometers (shallow wells) were placed across the valley bottoms of each watershed near the flume location. Soil moisture and soil temperature probes were installed at two locations within each watershed and placed at multiple soil depths.

All monitoring of weather, spring flow, channel flow, soil moisture and depth to water was done through satellite uplinks and data are available for viewing on the web site: http://ifpnet.com .

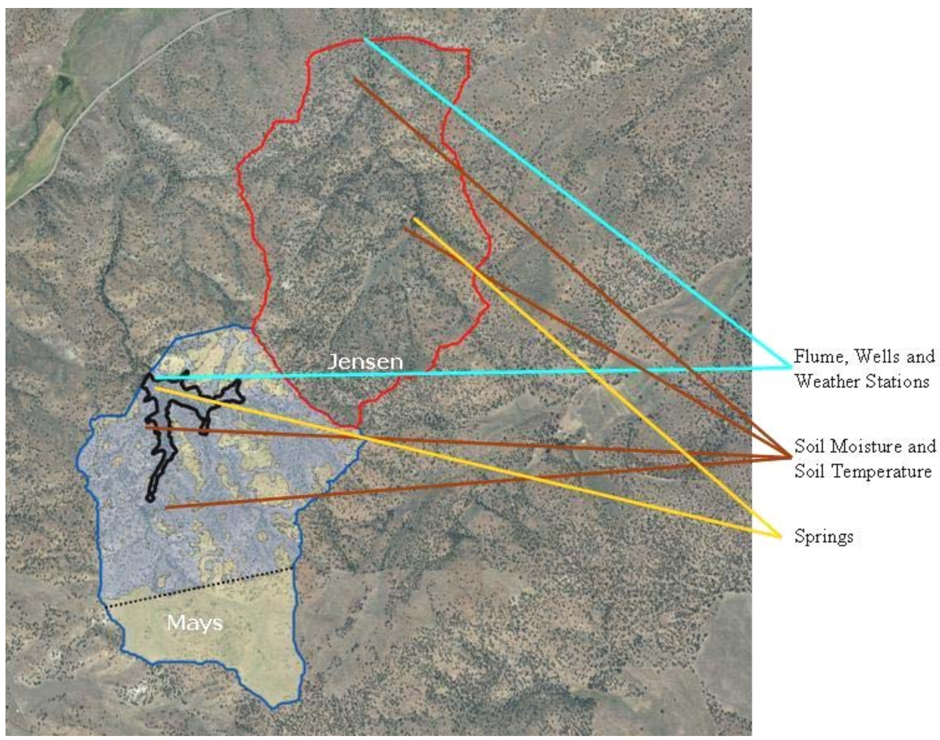


Figure 2. Location of monitoring stations in Jensen and Mays watersheds.

RESULTS

Spring Flow

The figure below and Table 1 illustrates the differences in output between the two springs and the differences between years. Spring flow is dependent on timing, type, and amount of precipitation. Base flow is least likely to be influenced by a recent precipitation event or snow melt period and is equivalent to late season flow. Late season flow is defined as the period between July and November.

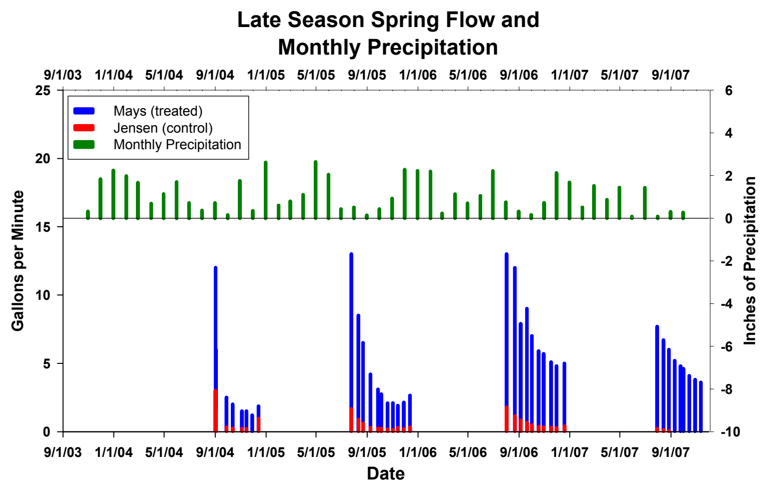


Table 1. T-Test for Spring Flow Data, lowest flow recorded (GPM). Data shows results for comparisons of late season flow (lowest flow recorded) between the two watersheds and the pre and post treatment years. The one tailed P-value is significant at alpha = .05.**

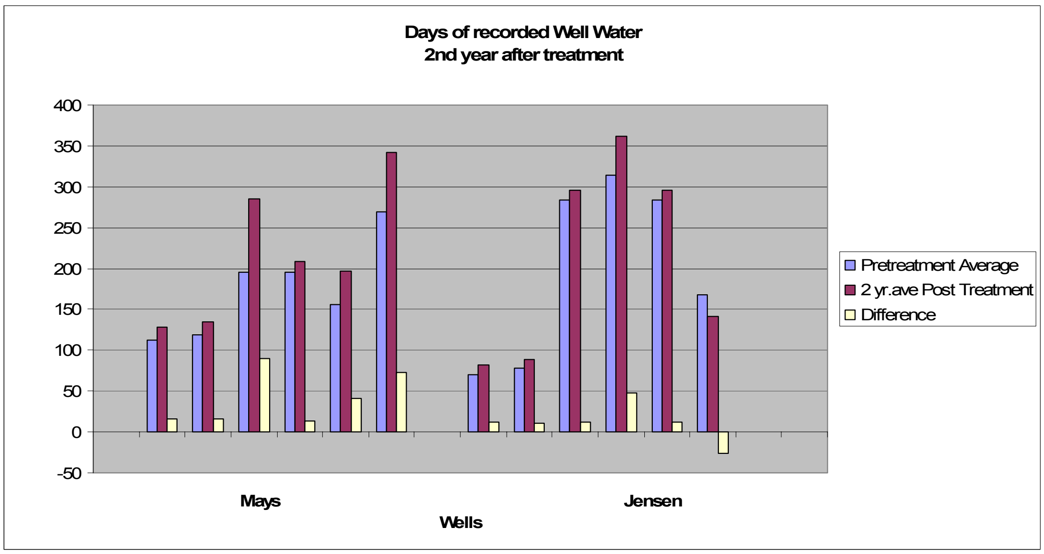
Treatment	Year	Watershed Mays	Watershed Jensen	Difference	Mean	Variance
Pre	2004	1.87	0.20	1.67		
Pre	2005	1.90	0.13	1.77	1.720	0.00500
Post	2006	4.80	0.23	4.57		
Post	2007	3.6	0.00	3.60	4.085	0.47045
		Difference			2.365	
Standard error	= 0.4875705;	t-test=4.8505805, One tailed P-value			0.0199857 **	

Table 2. Comparison of Average number of days of well water for the watersheds. Pre and post treatment years consist of 2 years each.

Watershed	Well	Pretreatment	Post treatment	Difference
Mays	1	112.5	128.5	16
	2	119.5	135	15.5
	3	195.5	285	89.5
	4	195.5	209	13.5
	5	156	197	41
	6	269.5	342.5	73
Jensen	1	70	82	12
	2	78.5	89	10.5
	3	283.5	296	12.5
	4	314.5	361.5	47
	5	283.5	296	12.5
	6	167.5	141	-26.5

Peizometer Wells

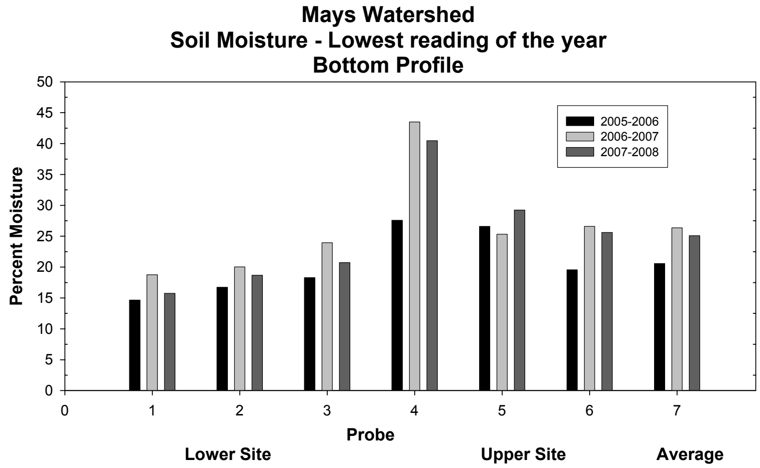
Well data provides insight to the timing or availability of subsurface water. The length of ground water availability could be an indicator of watershed function (Table 2). Increases in length would indicate an improved hydrologic condition (see figure below). A review of the data indicates that changes in the average number of days in which water was recorded in the wells increased in Mays watershed as a result of cutting the trees. Using a Wilcoxon rank test the wells in Mays post-treatment, recorded a greater increase in the number of days that water was recorded when compared to the control watershed (Jensen).



Soil Moisture

Observing the lowest readings of the year within each watershed illustrated the amount of "water savings" that was carried over from one year to the next. Evaluating the change in "water savings" over years helps us see if that change was associated only with precipitation, or if increases might have been due to the lack of deep-rooted vegetation (the cutting of the juniper). If it was due to the removal of deep rooted vegetation, then excess soil moisture could move through the soil profile and into sub-surface water storage and flow.

Individual probe readings were averaged by location within the soil profile and by site for each watershed (Figures next page). Analysis showed that the observed increase was significant for; the difference between 2006 and 2005, the average increase difference of 2006-2007 combined, and 2005 when comparing Mays with Jensen. Table 3 shows the results of this test for the combined years 2006-2007 compared to 2005.



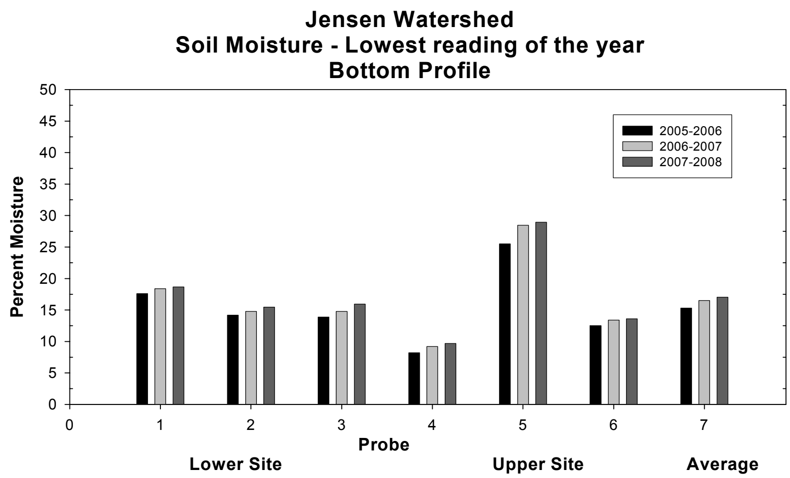


Table 3. Significance of end of year soil moisture accumulation post vs. pre treatment.

Year	Profile Location	P-value Difference
2006-07 vs. 05	Bottom (0.27 in)	.1002*
2006-07 vs. 05	Middle (0.18 in)	.1796
2006-07 vs. 05	Top (0.7 in)	.6132

Channel Flow

Channel flow in the two watersheds is ephemeral. These channels have flow only during periods of snow melt and extreme summer thunderstorm activity.

Comparisons of ephemeral channel flows or days of flow did not show a relationship to the treatment. In most years, recorded channel flow occurs during the spring and early summer months. In 1996 and 2004, total annual days of flow were greater than days of springtime channel flow, a result of late summer thunderstorms and early fall rain. In all years but one, Mays flowed longer than Jensen. Only in 1998 did Jensen flow longer when compared to Mays. In 2007, while length of flow was greater in Mays, Jensen's flow as measured in accumulated cubic feet per second was greater than Mays' flow.

Of special note in the observation of these systems was the winter of 2006, following the cutting of juniper in Mays. The snow pack, which began accumulating in December, 2005 was static at approximately 16 inches. December and early January rain events saturated the snow pack. As mentioned earlier, soil temperatures during this period did not drop below 32 degrees F for either watershed. Channel flow in Mays began on January 7, 2006. Flow was recorded through mid-June, 2006. In contrast, flow in Jensen did not begin until April 1, 2006 and ceased to flow by early May. During this period, all observations for both watersheds indicated that flow was generated exclusively from bank seepage and that no evidence of overland flow was observed for either watershed.

In contrast, during the winter of 2007, very little snow pack was accumulated. Bare ground in both watersheds accounted for 50–70% of the landscape, with snow accumulation areas measuring less than 6 inches. Soil temperatures in early February were approximately 22^{o} F. An early February storm produced a rain on snow event. Flow was recorded in both watersheds and the majority of channel flow originated as overland flow. Sediment movement was observed on the hill slopes and in the channels and sediment had to be removed from both flumes. The observations in 2006 and 2007 illustrate the high variability within these systems and the difficulty in connecting channel flow data to treatment effects, especially during the first 2 years following treatment.

Channel morphology

Channel morphology (the shape of the channel) was unique to each of the two watersheds at the time the study was initiated. The channel in Jensen can be described as being generally shallow, with less steep sides and a wider channel bottom. The channel could be characterized as U-shaped with the channel bottom controlled by rock. The channel depth is rarely more than 3 ft deep. This channel appeared to be influenced more by side-hill erosion processes as demonstrated by channel cross-sections being completely silted in due to side-hill sediment movement.

The channel in Mays can be characterized as being deeper and V-shaped with multiple head cuts found along its length. This channel bottom was not controlled by rock. Portions of the channel in Mays exceed 12 ft of depth. The channel in Mays also tends to support longer periods of flow through a greater expanse of its length than does the channel in Jensen. This channel appeared to be controlled more by in-channel processes (head cutting).

Time series analysis demonstrated greater variation of change in cross-sectional areas in the treatment versus the control watershed. The channel data from Mays tended to depict periodic, extreme soil movement events intermixed with periods of little to no soil movement, whereas, the channel data from Jensen showed more consistent soil movement with fewer extremes. This variation may be a product of the two channels being at different evolutionary stages relative to each other and thus indicating that channel recovery would be different for each watershed.

By acknowledging these inherent differences within each watershed, we may expect potentially different channel-forming processes. Long-term monitoring questions yet to be answered include how ephemeral channel processes work following vegetation treatments, and how recovery periods associated with vegetative treatments may differ based on channel morphology, associated vegetative conditions, and hydrologic processes.

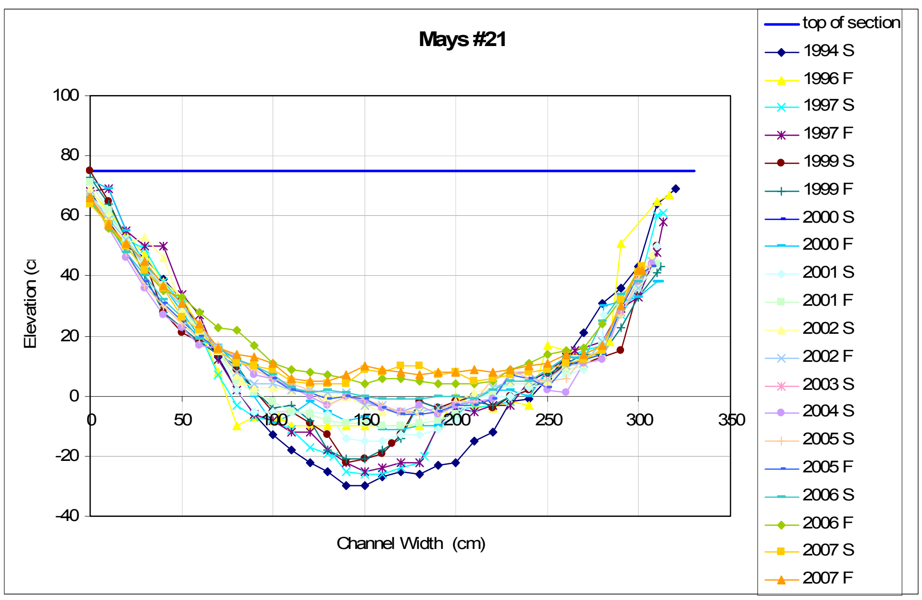


Diagram of channel morphological changes over time (1994-2007) using a single cross-section Mays 21.

The figure above demonstrates in cross-section the changes in channel morphology that have occurred since 1994 for the Mays watershed. Some channel cross sections show aggradations' or accumulation of sediments in the channel, trending upward towards a U-shaped channel. Other cross sections show channel degradation or continued downward movement of the channel bottom, creating a V-shaped channel.

MANAGEMENT IMPLICATIONS

A healthy, functioning watershed is one that captures, stores, and safely releases the precipitation that is delivered to the site. Land management decisions should include looking for ways to increase opportunities for precipitation to infiltrate into the soil profile, moving excess moisture into sub-surface storage and ground water, slowly releasing that water to minimize the risk of soil loss, and stabilizing channel beds and banks. Others have suggested that there would be no water yield increase as a result of vegetation manipulation (juniper cutting) in precipitation zones where annual precipitation was less than 4300 mm (17 inches). Any change to the water budget would only yield an increase in soil moisture, improving herbaceous vegetative production.

The 30-year average annual precipitation at Barnes Station (USGS weather station) located approximately 10 miles east of the study site is 3492 mm (13.8 inches). Precipitation over the last 4 years on the study site has ranged from 2781 mm (10.9 in., 80 percent of average) and 4491 mm (17.7 in., 129 percent of normal). Both the high and low precipitation years occurred during the post-treatment phase of the study.

A review of the data collected over the course of the last 13 years indicated that the cutting of post-settlement aged juniper has changed the water balance equation. Analysis of the first 2 years following treatment has shown that spring flow, ground water, and soil moisture have all increased when compared to pre-treatment levels. The comparisons of ephemeral channel flows did not show as clear a trend (data not presented here). This is likely because ephemeral flows tend to contribute more to ground water rather than ground water contributing to channel flow.

In the uplands, management implications suggest that with juniper removal, herbaceous vegetation can create a more uniform ground cover across the hillslope. Reduced bare ground results in increased infiltration opportunity and decreased soil erosion. Improved hydrologic function of the uplands can maintain site stability and fertility.

Within the riparian area, management implications point to the opportunity to increase spring flow for livestock, wildlife, and domestic use along with some mitigation of water diversion. Late season low flows often limit land management alternatives. Increasing flows by cutting juniper could partially offset this limitation. Changes in ground water may have downstream impacts such as adding to channel or perennial stream flows.

By combining the upland and riparian benefits of juniper removal, the system will begin to move toward a watershed that is functional in its ability to capture, store, and safely release water and provide a site that is productive and capable of being managed for sustainable use.

REFERENCES

Fisher, M.P., T. Deboodt, J. Buckhouse. 2008. Channel geomorphological changes and hillslope soil movement following juniper treatment on Camp Creek paired watershed study. Society of Range Management, 61st Annual Meeting, Louisville, Kentucky.

Gedney, D.R., D.L. Azuma, C.L. Bolsinger and N. McKay. 1999. Western juniper in eastern Oregon. General Technical Report PNW-GTR-464. USDA Forest Service Pacific Northwest Research Station, Portland Oregon. 53 p.

Hibbert, A.R. 1983. Water yield improvement potential by vegetation management on western rangelands. Water Resources Bulletin 19:375-381.

Larsen, R.E. 1993. Interception and water holding capacity of western juniper. PhD. Dissertation. Oregon State University, Corvallis, OR. 172 p.

Runoff and Erosion after Cutting Western Juniper

Frederick B. Pierson, Jon Bates, Tony Svejcar, and Stuart Hardegree

INTRODUCTION

We used rainfall and rill simulation techniques to evaluate infiltration, runoff, and erosion on cut and uncut western juniper treatments. Research from pinyon-juniper watersheds in the southwest demonstrates a strong relationship between vegetation cover and soil erosion by wind and water. The purpose of this study was to quantify hydrologic changes associated with vegetation recovery 10 years after western juniper control in eastern Oregon. Specific objectives were to measure changes in surface runoff and rill erosion as a function of rainfall intensity and to assess surface soil and vegetation factors that are influencing hillslope hydrology and erosion.

METHODS

The study site was on Steens Mountain in southeastern Oregon. Elevation at the site is 1575 m and aspect is west facing with a 10% slope. The site was dominated by post-settlement western juniper woodland (Phase III) Juniper canopy cover averaged 26.5% and tree density averaged 283 trees/ha. Shrubs were eliminated from the site. Herbaceous cover averaged 5.5 %. Bare ground and rock in the interspace was about 95%. Treatments consisted of removing juniper by cutting and allowing the understory vegetation to recover for 10 growing seasons and an uncut juniper woodland control. Eight 2-ac blocks were established. All trees on half of each block were cut in 1991. Simulations were applied to eight cut plots and eight woodland plots.

Surface hydrology and understory vegetation response variables were compared between cut and uncut juniper woodlands. Observation indicated that most runoff moved through interspace areas: therefore, study plots were placed in interspaces between cut or uncut trees. Simulated rainfall was applied at a target rate of 2 in. hr^{-1}, for 1 hour, to 37.5 yd^{-2} plots using eight stationary sprinklers in June 2001. Runoff and sediment were collected during the simulation period. One hour following the rainfall simulation, a flow regulator was used to measure runoff volume, sediment concentration, and flow velocity in rills.

RESULTS

Vegetation

Cutting junipers increased total vegetation cover (canopy and basal) and litter cover (Table 1). Cover was about four times greater in the cut versus woodland treatment. Bareground in woodland intercanopy zones was 2-3 times higher than the cut treatment. Rooting characteristics differed slightly between treatments (Table 2). In the juniper woodlands, roots originated primarily from juniper and in the cut treatment roots were primarily composed of perennial grasses. Root mass was greater in the woodland, though, root length and root length density were greater in the cut treatment. Root length and root-length density data indicate that the cut treatment had more fine roots than the juniper woodland. This resulted in increases in roughness and aggregate stability found in the cut treatment compared to the uncut woodland.

Runoff

Woodlands rapidly produced large amounts of runoff while cut plots produced almost no runoff (Figure 1, Table 3). All woodland plots began to runoff within 16 min following the start of rainfall. Four plots began to runoff after only 2- 4 min. Only two cut plots generated any runoff during the 1-hourainfall simulation. One cut plot began runoff at 31 min. and the other at 43 min. The woodland plots were on average 82% ponded (surface saturated) while the cut plots were only 30% ponded.

The application of a 2-year return period thunderstorm equivalent on four juniper plots (50% of the woodland area) produced runoff, whereas no cut plots produced any runoff. With the application of a 4-year return period storm, seven juniper woodland plots (88% of the area) produced runoff; no cut treatment plots produced any runoff. A 50-year return period storm had to be applied before two cut plots (25% of the cut area) began to produce runoff whereas all eight of the juniper plots produced an average of 4 mm of runoff.

Sediment Yield

Sediment yield was orders of magnitude higher for the juniper woodland (1,052 lb/acre) compared to the cut treatment (12.6 lb/acre) (Fig. 1, Table 3). The sediment to runoff ratio, a measure closely associated with soil erodibility, was two times greater in the uncut woodland than the cut treatment. This indicates that soil particles were more easily detached on woodland sites compared to areas in the cut treatment.

Table 1. Ground cover (%) and canopy cover (%) in the intercanopy zones between trees for juniper woodland and juniper-removed treatments, Steens Mountain, Oregon, 2001. Upper case letters denote significant treatment differences for individual ground cover components between treatments. Lowercase letters denote significant treatment differences for individual canopy cover components.

	Canopy Cover	
	Juniper Woodland	Juniper Removed
Perennial Grass	1.1 ± .05a	12.7 ± 2.6b
Annual Grass	0.02 ± 0.01a	2.9 ± 0.8b
Perennial Forb	0.8 ± 0.4a	1.7 ± 0.5b
Annual Forb	1.2 ± 0.1	1.4 ± 0.2
Shrub	0.0 ± 0.0a	1.5 ± 1.1b
Vegetation Total	5.6 ± 1.2a	23.2 ± 3.1b
Litter	8.1 ± 1.4a	18.8 ± 2.2b
Rock	6.3 ± 2.4a	4.6 ± 1.8b
Bare Ground	79.9 ± 7.0a	53.3 ± 9.5b

Rill Dynamics

Rill discharge was 3 to 7 times higher for the juniper woodland compared to the cut treatment for all inflow rates tested (Table 4, Fig. 2). Sediment to runoff ratios were significantly higher for the juniper plots, indicating higher rill erosion compared to the cut plots. Rills in the cut areas remained in more of a transport-limited state due to low rill discharge rates. The number of rills was nearly 50% greater and the width of flow within each flow-path was higher in the juniper woodland compared to the cut areas. Water velocity was twice as high in the juniper plots.

Table 2. Hillslope characteristics for juniper woodland and juniper removed treatments, Steens Mountain, Oregon, 2001. Lower case letters denote significant treatment differences.

	Juniper Woodland	Juniper Removed
Slope (%)	18.5± 2.0	19.2± 1.3
Random Roughness (m)	0.024±0.007a	0.036±0.012b
Bulk Density 0-3 cm ($g\ cm^{-3}$)	1.51± 0.10	1.52± 0.09
Bulk Density 3-6 cm ($g\ cm^{-3}$)	1.46± 0.06	1.51± 0.06
Sand (%)	46.0± 7.6	45.2± 5.3
Silt (%)	38.8± 4.6	37.5± 3.9
Clay (%)	15.2± 4.1	17.3± 3.4
Organic Carbon (%)	1.82± 0.51	1.94± 0.71
Aggregate Stability (%)	44.8±10.4a	62.7±8.6b
Root Mass ($g\ m^{-3}$)	214 ± 20 a	130 ± 22 b
Root Length (cm)	11.8 ± 10.5	14.3 ± 30.9

Table 3. Comparison of cumulative runoff and sediment yield between juniper woodland and juniper removed treatments for different return period storms, Steens Mountain, Oregon. Upper case letters denote significant differences between treatments for runoff. Lowercase letters denote significant differences for sediment yield.

Time (min)	Rainfall (mm)	Storm Return Period (y)	Runoff (mm)		Sediment Yield ($lb\ ac^{-1}$)	
			Juniper Woodland	Juniper Removed	Juniper Woodland	Juniper Removed
5	4.45	2	0.11	0.00	7.1a	0.00b
10	8.89	4	0.53A	0.00B	33.3a	0.00b
15	13.36	8	1.15A	0.00B	70.9a	0.00b
30	26.67	50	3.93A	0.00B	265.0a	0.0b
60	53.34	100+	13.47A	0.96B	1052.5a	12.6b

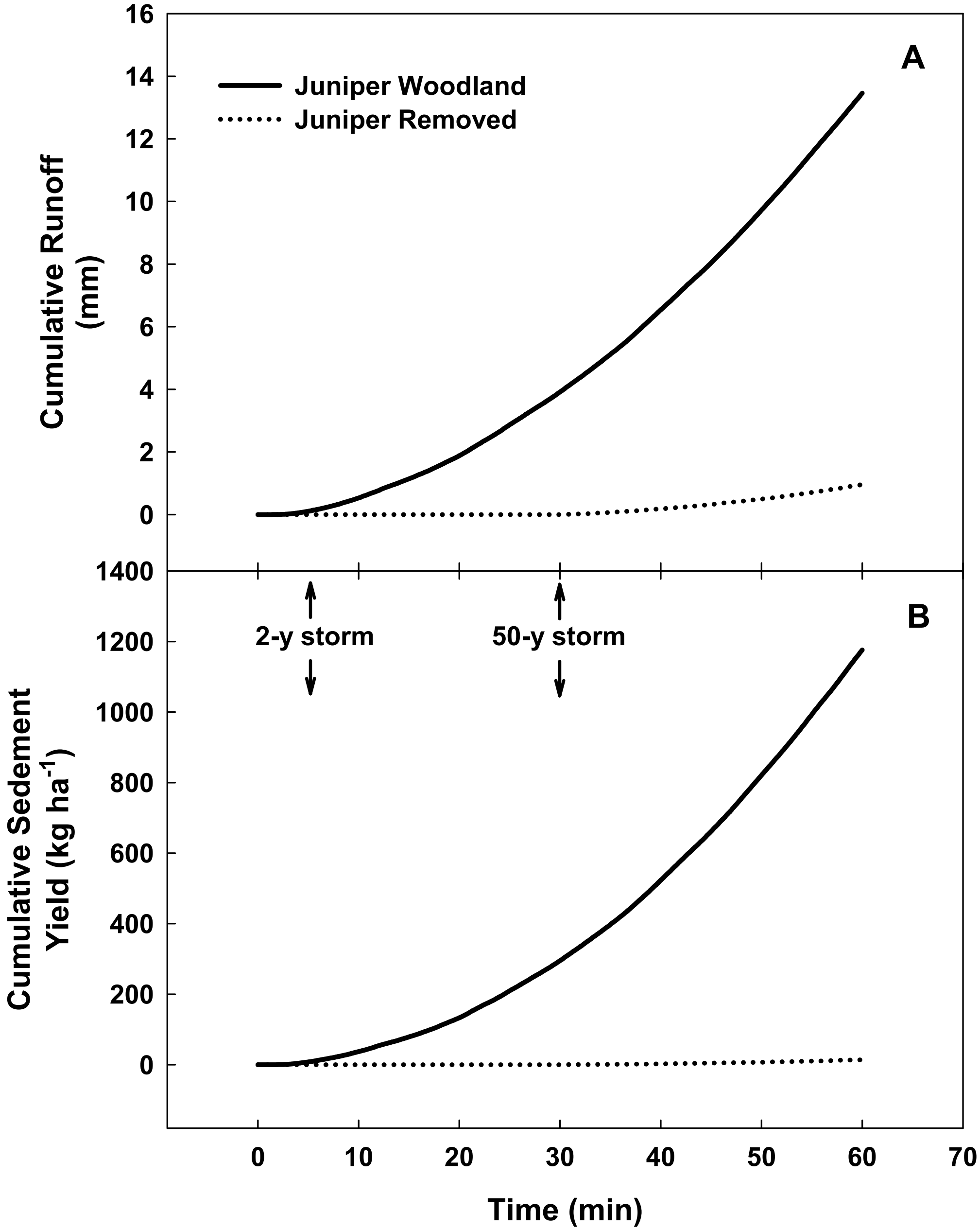


Figure 1. Average cumulative runoff (A) and sediment yield (B) for juniper woodland and juniper removed treatments (n=8), Steens Mountain, Oregon. Rainfall was applied at 2 inches/hr (100 year storm event). Other storm events are shown on the graphs.

Table 4. Rill flow characteristics by rill inflow rate for juniper woodland and juniper removed treatments, Steens Mountain, Oregon, 2001.

	Juniper Woodland			Juniper Removed		
Inflow Rate (l min^{-1})	7	12	15	7	12	15
Cumulative Inflow (L)	120	264	444	120	264	444
Cumulative discharge (L)	53.3*	154.3*	289.9*	7.8	35.4	82.5
Cumulative sediment (g)	114.2*	368.5*	626.8*	4.3	35.0	85.3
Sediment/Runoff (g L^{-1})	2.58	2.49	2.21*	-	-	0.75
Number of flow paths	2.6	2.7	2.9*	-	-	2.1
Flow velocity (m s^{-1})	-	0.098	0.110*	-	-	0.067
Flow depth (mm)	7.7	8.3	8.5	-	-	8.3
Flow path width (m)	0.23	0.24	0.26	-	-	0.22
Total flow width (m)	1.11	1.10	1.13	-	-	0.79

* Treatment differences are significant for associated inflow rate.

CONCLUSIONS

This study highlights the importance of maintaining understory vegetation and litter cover when managing western-juniper encroachment. Management practices that maintain adequate ground cover reduces soil loss and retains site productivity.

The hydrologic impacts of western-juniper in this study are consistent with studies that have linked changes in infiltration, runoff, and erosion to declines in understory vegetation and surface litter that result in larger, more inter-connected areas of bare ground. Large patches of inter-connected bare ground provide better opportunity for runoff to concentrate into rills with high flow velocity, high erosive force, and sediment transport capacity.

The high rill erosion rates found in untreated juniper plots in this study were a result of the increase in velocity of water moving along a greater number of flow paths. Less ground cover and increased bare ground in the juniper woodland provided less resistance to water moving over the soil surface. Overland flow could then pick up more speed and thus, energy for detachment and transport of soil particles in the flow paths. This coupled with significantly lower infiltration capacity and aggregate stability in the juniper woodland resulted in greater rill and interrill discharge rates and sediment concentrations.

Cutting the juniper and allowing the site to recover for a 10-year period was very successful at restoring the site to a hydrologically stable condition. Surface soil cover was restored and infiltration capacity increased sufficiently to protect the site from even large thunderstorms.

When runoff was generated in the cut areas, the improved surface cover conditions reduced the amount and velocity of overland flow, thereby dramatically reducing rill erosion rates.

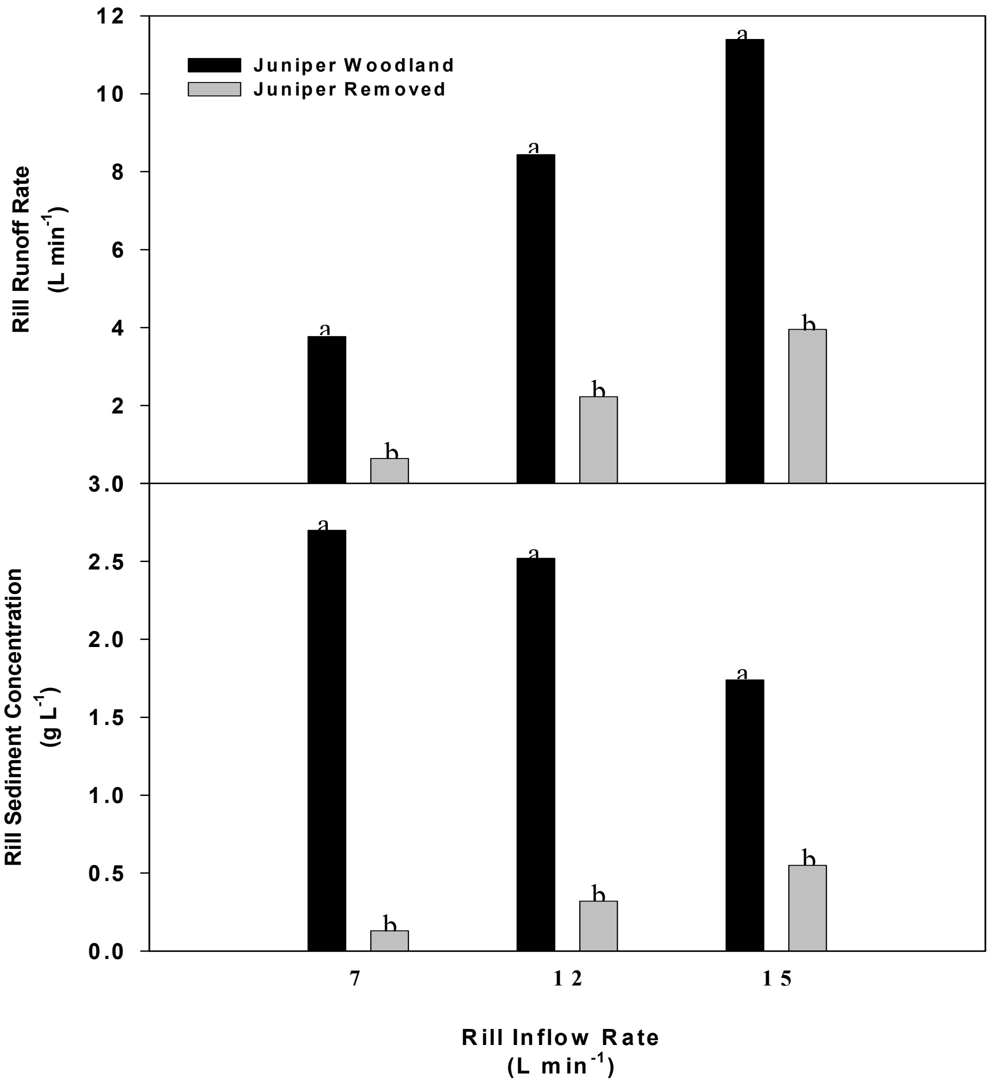


Figure 2. Average rill discharge rate (A) and sediment concentration (B) by rill inflow rate (in liters) for juniper woodland and juniper removed treatments (n=8) on Steens Mountain, Oregon, 2001. Values for the same rill inflow rate with different letters are significantly different (p<0.05). Runoff and sediment concentration are all much greater in the woodland compared to the cut treatment.

Restoration of Quaking Aspen Woodlands Invaded by Western Juniper

Jon D. Bates, Rick Miller, and Kirk W. Davies

INTRODUCTION

Quaking aspen woodlands are important plant communities in the interior mountains of the western United States. Occupying relatively small areas within vast landscapes, aspen woodlands provide essential habitat for many wildlife species and contain a high diversity of understory shrub and herbaceous species. Western juniper woodlands are rapidly replacing lower elevation (<6800 ft) quaking aspen stands throughout the northern Great Basin. Fire exclusion has resulted in juniper encroachment or replacement of aspen woodlands the past 100 years.

The purpose of this research was to evaluate two juniper control treatments for restoring aspen stands in eastern Oregon using selective cutting and prescribed fire. The two juniper control treatments involved cutting one-third of mature juniper trees followed by 1) early fall burning (FALL); or 2) early spring burning (SPRING). Because of a lack of fine fuels and relatively high fuel moisture contents, selective cutting of juniper was done to increase surface fuels (0-6 ft) to carry fire through the aspen stands, kill remaining juniper, and stimulate aspen regeneration. Specific objectives were to: 1) test the effectiveness of treatments at removing juniper from seedlings to mature trees, 2) measure treatment effectiveness at stimulating aspen recruitment, and 3) evaluate the response of shrub and herbaceous understories to treatment.

METHODS

The study site was located in Kiger Creek Canyon on Steens Mountain, southeastern Oregon. The two western juniper control treatments involved cutting one-third of the mature juniper trees followed by: 1) early fall burning (FALL); or 2) early spring burning (SPRING). Treatments were located next to untreated woodlands (CONTROL). Trees were cut in winter and spring, 2001. The FALL treatment was burned in mid-October, 2001. The SPRING treatment was burned in mid-April, 2002. Sampling included measurement of cover and density of juniper, aspen, shrubs, and herbaceous species and understory diversity. Sites were measured in June-July 2000 and 2002-2006.

RESULTS

Cut and Fall Burn

The FALL treatment was a severe-stand replacement fire, resulting in greater plant mortality and more open spaces for colonization by new individuals when compared to the SPRING treatment. The greater disturbance in the FALL treatment favored aspen recruitment and growth. Burning eliminated remaining juniper trees and seedlings (Figs. 1

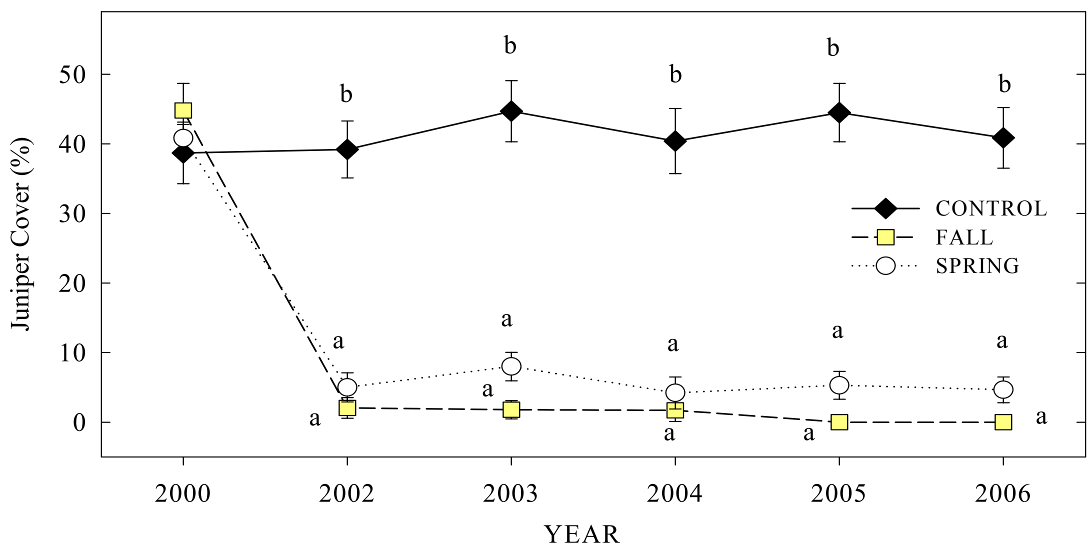


Figure 1. Juniper cover in aspen stands prior (2000) to and after treatments, Kiger Canyon, Steens Mountain Oregon. Letters denote significant differences among treatments and untreated Control.

& 2) and stimulated a 6-fold increase in aspen suckering (5,420 stems/acre) (Figs. 3 & 4), but resulted in a severe reduction in herbaceous cover and loss of perennial bunchgrasses.

Herbaceous cover in FALL was less than the SPRING and was primarily composed of weedy annuals (native and non-native) (Fig. 5). In 2006, cheatgrass made up nearly 60% of total herbaceous cover.

Cut and Spring Burn

The SPRING treatment was a less severe fire that thinned the overstory and resulted in a substantial increase in herbaceous cover and diversity. Eighty percent of the mature juniper trees that remained after cutting were killed, however, 50% of juniper juveniles survived (juveniles exceed 300 trees/acre) (Fig 1 & 2).

Aspen suckering in the SPRING treatment increased 3.5-fold (2,985 stems/acre) by the fifth year post-fire (2006) (Figs. 3 & 4). The SPRING treatment has prolonged aspen site occupancy but the presence of juniper will result in co-dominance of the overstory by aspen and juniper within 30 years.

Herbaceous cover increased 330%, no mortality of bunchgrasses occurred, and the number of species observed increased by 50% by the fifth year after fire (Fig. 5). It is estimated that livestock forage increased about 10-fold. Herbaceous composition was

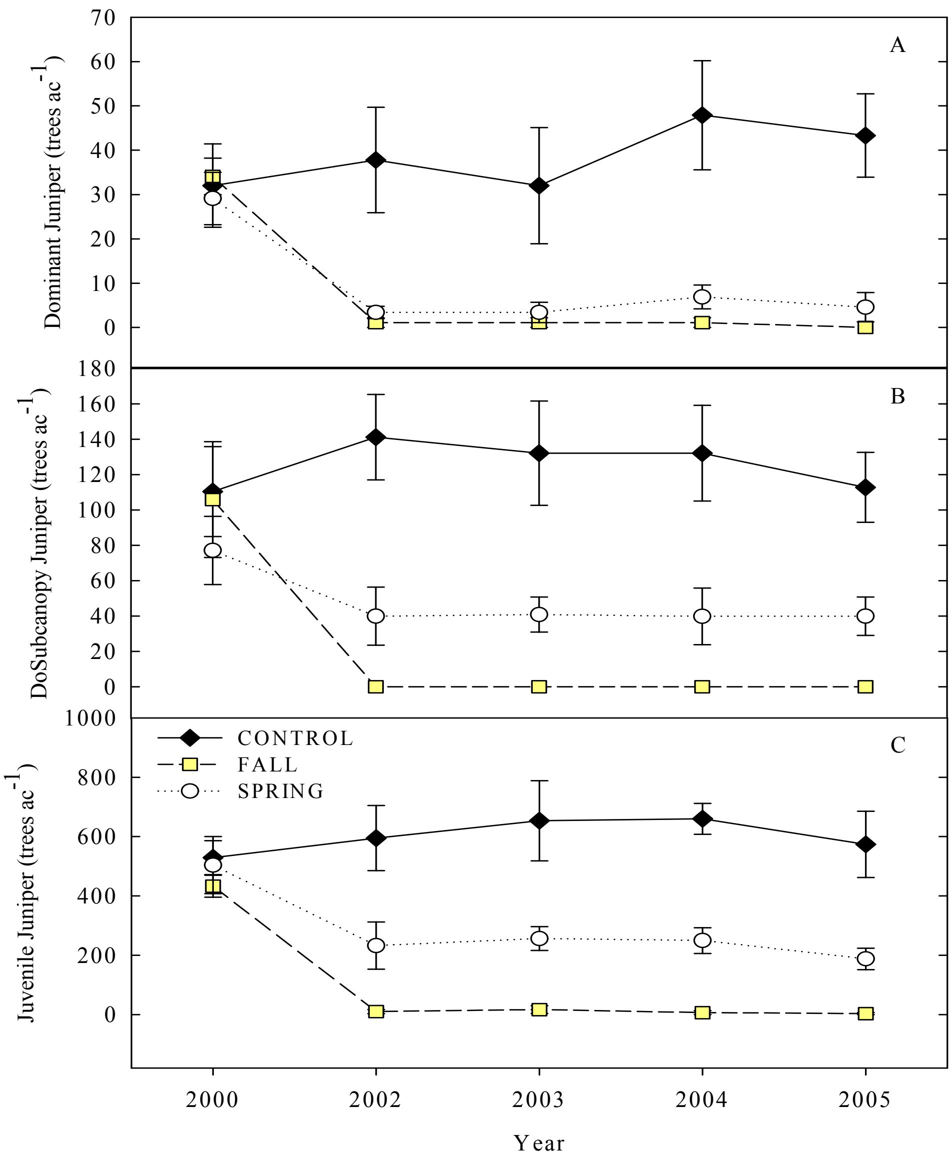


Figure 2. Juniper tree densities in aspen stands prior to (2000) and after treatments, Kiger Canyon, Steens Mountain Oregon; A) dominant juniper; B) sub-canopy juniper; and C) juvenile trees (< 3 ft). The CONTROL is greater than the treatments for all categories. Juvenile junipers are greater in the SPRING than the FALL.

primarily composed of native grasses and forbs. Perennial forb diversity was highest in the SPRING treatment.

CONCLUSIONS

Cut and Fall Burn

Cutting combined with fall fire was the most effective method at removing remaining juniper and stimulating greater aspen suckering. The effectiveness of the treatment at removing juniper indicates that aspen will dominate the overstory the next 80-100 years. The cutting of

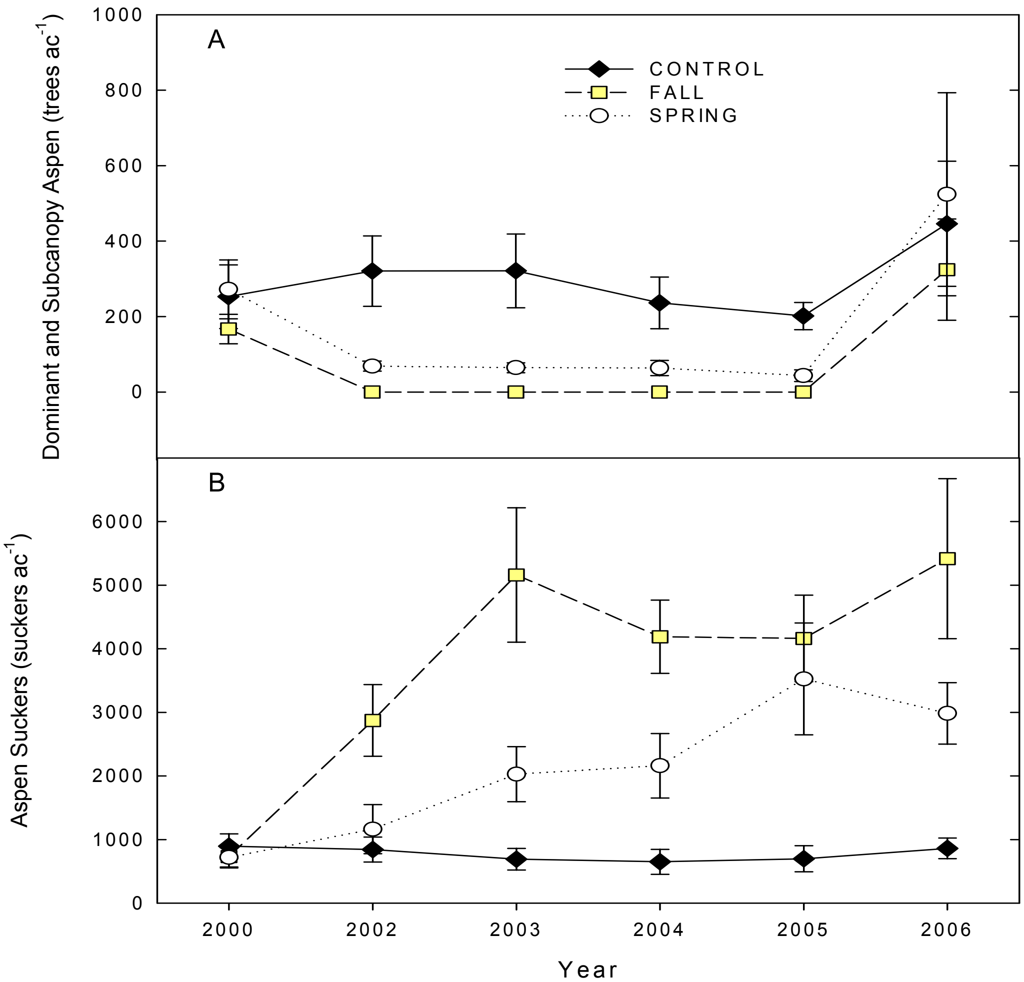


Figure 3. Aspen densities prior to and after treatments, Kiger Canyon, Steens Mountain Oregon; A) dominant and subcanopy aspen; B) aspen suckers (< 2 in diameter at 3 ft). The CONTROL was greater than the treatments for dominant and subcanopy aspen until 2006. In 2006 many aspen suckers In FALL and SPRING treatments began to be categorized as subcanopy because stem diameters exceeded 2 inches.

one-third of the overstory juniper was more than adequate to eliminate remaining live junipers with FALL fire treatment. This suggests that cutting levels could potentially be reduced when combined with fall fire.

Fall fire severely impacted the understory and reseeding of herbaceous perennials should be considered. Cut trees increase heat fluxes into soils and elevate mortality of perennial species. Native perennial forbs and grasses were largely eliminated with the fall fire. In these lower elevation aspen stands, non-native weeds appear to be of concern in early succession as they rapidly increase before native perennials can reestablish.

What has been surprising is a steady increase of cheatgrass in the FALL treatment. Cheatgrass is unlikely to persist as Kentucky bluegrass that survived the fire has slowly increased and will likely reoccupy treated sites.

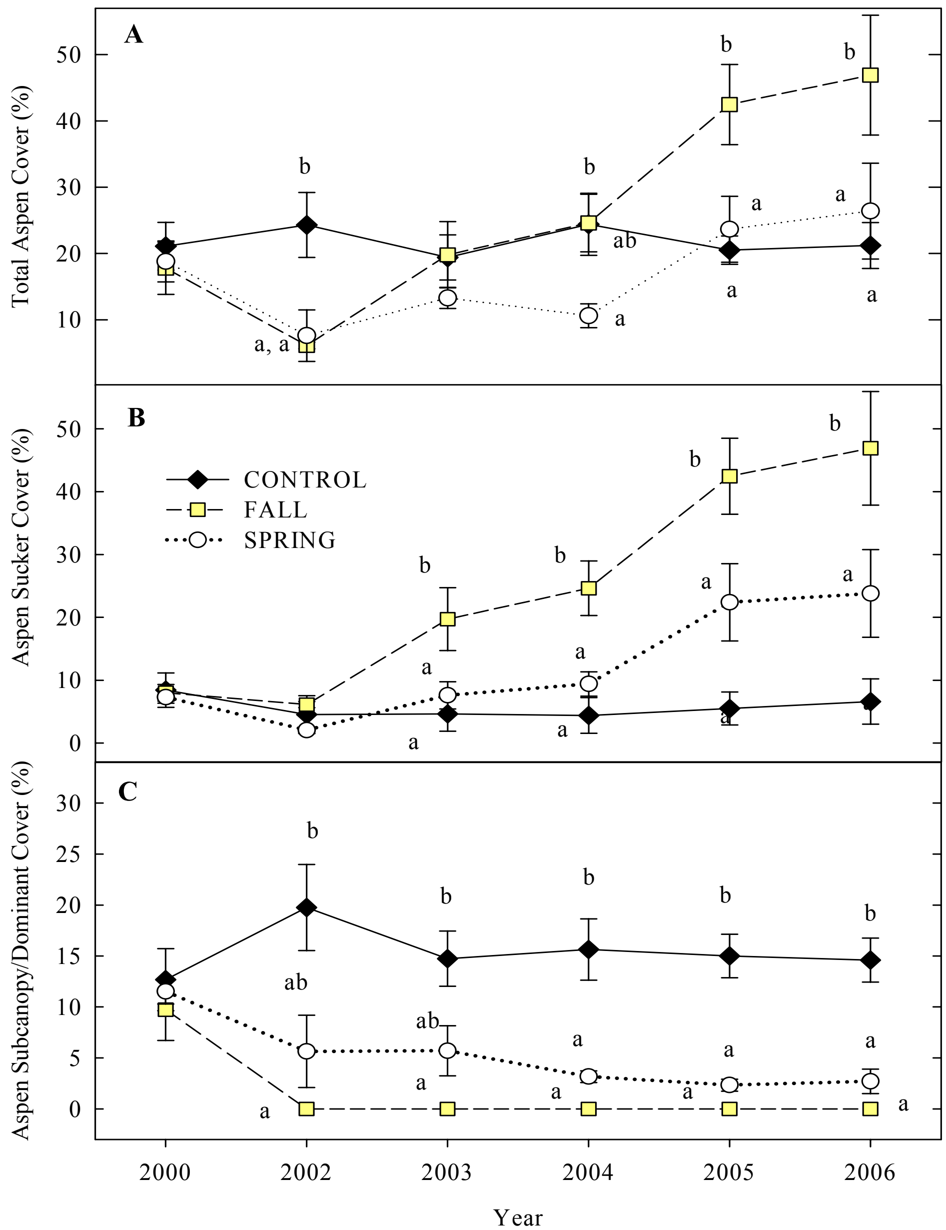


Figure 4. Aspen cover prior to and after treatments, Kiger Canyon, Steens Mountain Oregon; A) total aspen cover; B) aspen sucker cover; and C) dominant and subcanopy aspen. Letters denote significant differences among SPRING and FALL treatments and untreated CONTROLS.

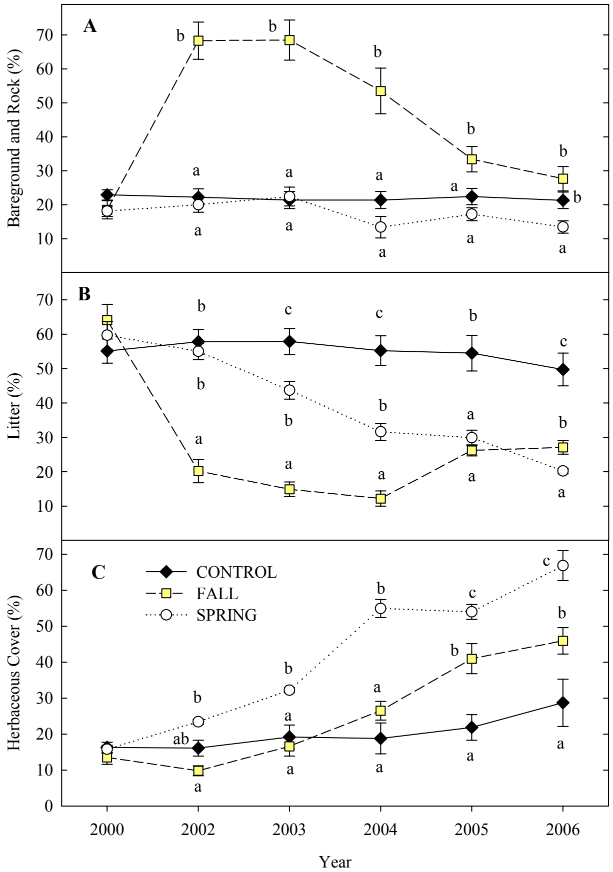


Figure 5. Ground cover prior to and after treatments, Kiger Canyon, Steens Mountain Oregon; A) Bareground; B) Litter; and C) Herbaceous cover. Letters denote significant differences among SPRING and FALL treatments and untreated CONTROLS.

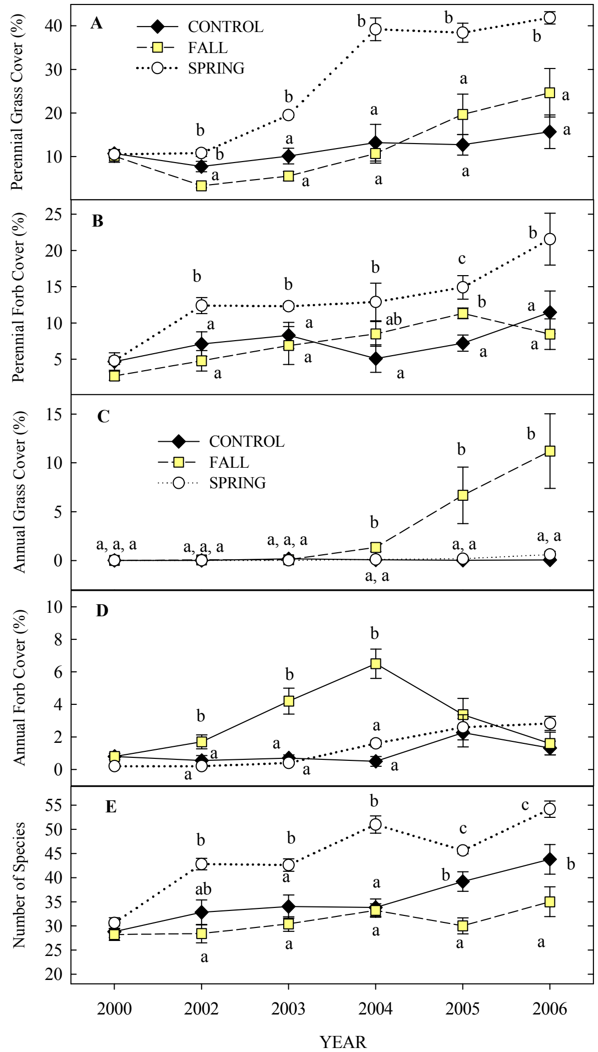


Figure 6. Herbaceous functional group cover and herbaceous species identified prior to and after treatments, Kiger Canyon, Steens Mountain Oregon; A) Perennial Grasses; B) Perennial Forbs; C) Cheatgrass; D) Annual forbs; E) Species numbers. Letters denote significant differences among SPRING and FALL treatments and untreated CONTROLS. Perennial grasses and forbs and species numbers are all greatest in the SPRING. Cheatgrass makes up about 25% of total herbaceous cover in the FALL. Also of note is that species numbers has been lowest in the FALL the past 2 years (2005-2006).

Cut and Spring Burn

This treatment can be considered only a temporary interruption of the development to juniper woodland. The gaps created by the cutting and fire disturbance will provide juniper saplings and seedlings with the opportunity to reoccupy the SPRING sites. However, if the management objective is to rapidly increase the herbaceous component and moderately increase aspen suckering, spring burning is recommended. Spring burning may also be useful in aspen communities where the understory is depleted and managers desire a more rapid recovery of this vegetation group.

With spring burning, follow-up management will be necessary to remove juniper that are missed in initial treatments to prevent early return and domination by juniper. Given growth rates of juniper, these stands could be re-dominated by juniper in about 60-80 years.

When sites are burned in spring (or winter) preparatory cutting levels should probably be increased above 50% to increase chances of remove a higher percentage of junipers, both mature and juvenile trees, by fire. This level of cutting would likely not impact the understory negatively when the site is burned as long as soils and surface litters are frozen and or at field capacity (litter in contact with the ground); and herbaceous vegetation is largely dormant.

An advantage of spring burning is that the fire can be confined to the treatment area without much risk of escape. This treatment would be useful in other forested systems (e.g Ponderosa Pine, other encroaching conifer species) and in stands adjacent to areas of management concern (e.g. Mountain big sagebrush habitat, riparian zones, structures, residential areas etc.). For example, it may be desirable to protect areas, particularly sagebrush grassland, to avoid negative impacts to wildlife dependent on these communities.

ACKNOWLEDGEMENTS

Thanks to the Bureau of Land Management-Burns District for providing the opportunity to conduct the study and applying the fall burn treatment. Special recognition is for Jim Buchanan of Burns-BLM for his work on the project. Fred Otley and family were most generous in providing use of their summer cabin during sampling periods. Many student summer range technicians assisted in the collection of field data and ARS range technicians (Claire Poulson and Lori Zeigenhagen) assisted in the spring fire applications.

REFERENCES

Bates J.D., R. Miller, and K. Davies. 2006. Restoration of aspen invaded by western juniper. Range Ecology and Management 59:88-97.

Wall, T. R.F. Miller, and T.S. Svejcar. 2001. Juniper encroachment into aspen in the northwest Great Basin. Journal of Range Management 54:691-698.

A Generalized Model for Estimating Biomass and Fuel Loads for Western Juniper

Jaime Ratchford, Breanna Sabin, Andrew Tierney, Rick Miller, and Paul Doescher

INTRODUCTION

Western juniper has significantly expanded its range since the late 1800's and currently occupies 9 million acres in eastern Oregon, southwestern Idaho, and along the northern border of California and Nevada. The conversion from shrub-steppe to western juniper woodland shifts above-ground biomass from primarily one composed of shrubs and grasses to dominance by woody vegetation. This shift in biomass also entails a major change in the abundance and structure of ecosystem fuel loads. This shift in fuel load arrangement and composition probably decreases the potential for fire ignition because of a lack of fine fuels. However, because of the changes in fuel load characteristics, once woodlands are ignited they burn with higher intensity with the potential for more severe effects to residual understory vegetation and soil resources. Quantifying the changes in above-round plant biomass in invasive western juniper woodlands can provide land managers and researchers a better understanding of plant community and landscape dynamics. In addition, recent interest in harvesting western juniper for commercial energy generation requires a method that can estimate tree biomass across watershed and land management units.

Procedures for determining biomass and fuel loads in woodland systems are poorly developed and can be tedious and expensive. Biomass studies have produced allometric equations that can accurately estimate and inventory the biomass of western juniper at a local scale. Because past biomass equations are derived from a limited number of sites, it is unknown if these estimates can be used to scale up to larger land areas or landscapes. Developing estimates of juniper biomass at larger scales is needed to quantify fuel characteristics and potential energy resources.

The objective of this study was to develop allometric models of western juniper that would estimate tree biomass at multiple scales, including individuals, stands, or landscapes. To do this we selected three widely separated locations where we evaluated the relationship among different dimensional measures with above ground dry-weight of western juniper.

STUDY AREA

We selected three sites that were representative of invasive western juniper woodlands (Fig. 1). The predominate vegetation on these sites were western juniper, mountain big sagebrush, Idaho fescue, and Sandberg bluegrass. These sites were in phase II of the woodland succession where intraspecific competition is minimal, the overstory canopy is still open and the dominant native understory vegetation is still intact. The potential plant associations at all sites were mountain big sagebrush/Idaho fescue although annual precipitation and soils varies among the three study locations.

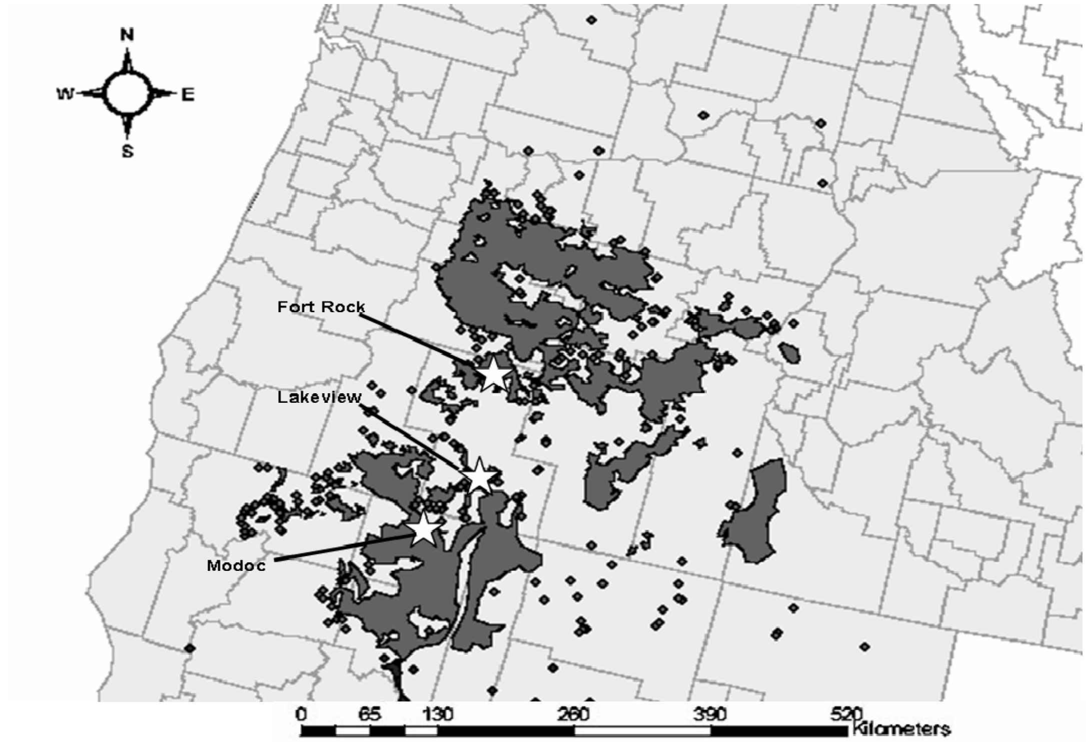


Figure 1. Distribution of western juniper woodlands and the locations of the three research plots.

The most northern site was located near Fort Rock, Oregon. It is considered to be in the Humboldt Ecological Province. Soils are well drained and shallow to moderately deep. They are clayey, semetitic, frigid, shallow, Vitritorrandic Durixerolls. The mean annual precipitation varies between 9-12 inches. Temperatures range from an average minimum of 24 °F in January to an average maximum of 86 °F in July with an average annual temperature range of 45 to 55 °F. The elevation was approximately 5,600 ft.

The second site was located near Lakeview, Oregon. It is also considered to be in the Humboldt Ecological Province, but is a cooler and wetter site than the Fort Rock site. Soils are well drained and moderately deep. They are fine, smectitic, frigid Vertic Palexerolls. The mean annual precipitation varies between 14-18 inches. The temperatures range from an average minimum of 20 °F in January to an average maximum of 84 °F in July with an average annual temperature range of 45 to 47 °F. The elevation was approximately 4,700 ft.

The third site was located in the Modoc National Forest near Tule Lake, California, and is in the Klamath Ecological Province. Soils are well drained and shallow to moderately deep. They are fine loamy to loamy, mixed, mesic, Lithic and Pachic Argixerolls. Annual precipitation varies between 16 – 20 inches. Average annual temperatures range from 43 to 45 °F and the elevation is approximately 5,000 feet.

METHODS

At each ecological site one, five-acre plot was established. Across the entire plot crown and trunk measurements were recorded for all trees over 9 ft tall. Measurements included live crown height, tree height, crown diameter, and basal diameter. To ensure the usefulness of the model we selected canopy area as the predictor of biomass because tree canopy area is easily obtained in the field and can be estimated using aerial photography. Crown area was calculated using the following equation:

$$\text{Canopy area} = \pi[(\text{crown diameter}_1 + \text{crown diameter}_2/4)]^2$$

A sub-sample of the trees measured was selected for destructive sampling at each of the three sites (n = 99). Trees were felled, all main branches (those originating from the main trunk) were removed from the main stem, and sectioned into manageable pieces (25 to 50 lbs). All portions of the tree were separated into four size classes based on standard fuel classes: 1-hr (<0.25 in), 10-hr (0.25 – 1.0 in), 100-hr (1.0 -3.0 in) and 1000-hr (> 3.0 in). Samples from each size class were placed separately into a tarp and weighed using a load scale. Sub-samples were collected for each size class of each tree at each site. These sub-samples were oven dried at 140°F to determine field moisture content. The moisture content was determined by weighing each sample daily or weekly until a constant weight was reached. Using the moisture content of the sub-sample for each size class the field wet weights were converted to dry weights.

RESULTS and DISCUSSION

A total of 99 western juniper trees were destructively sampled across all three sites. Western juniper canopy cover ranged from 3% to 63% with the lowest cover being found at the Fort Rock site and the highest cover at the Lakeview site. We found tree canopy area was a reliable predictor of total tree biomass and biomass of all four fuel classes (Table 1). Total biomass for a single juniper ranged from 26 to 1,474 lbs across the three sites. For every square foot of canopy cover western juniper gains 2.25 lbs of biomass. Figure 2 predicts western juniper biomass (lbs/ac) as percent canopy cover. At the landscape or stand scale, total juniper biomass increases by 9,801 lbs for every acre of cover (Table 2). Of the four fuel classes, the majority of biomass was in the 1-hour fuels class, which was composed mostly of tree foliage (Fig. 3).

Table 1. The relationships between canopy cover and the above ground biomass of the four fuel classes and the total biomass of western juniper. Dry weight is expressed in lbs (y). Juniper canopy area is expressed in square feet (x).

Fuel Class Category	Equation	R^2	*P*
1-hr	$y = 41.43 + 0.46x$	0.55	<0.0001
10-hr	$y = 9.83 + 0.28x$	0.70	<0.0001
100-hr	$y = -30.51 + 0.68x$	0.78	<0.0001
1000-hr	$y = -19.3 + 0.83x$	0.75	<0.0001
Total	$y = 1.5 + 2.25x$	0.81	<0.0001

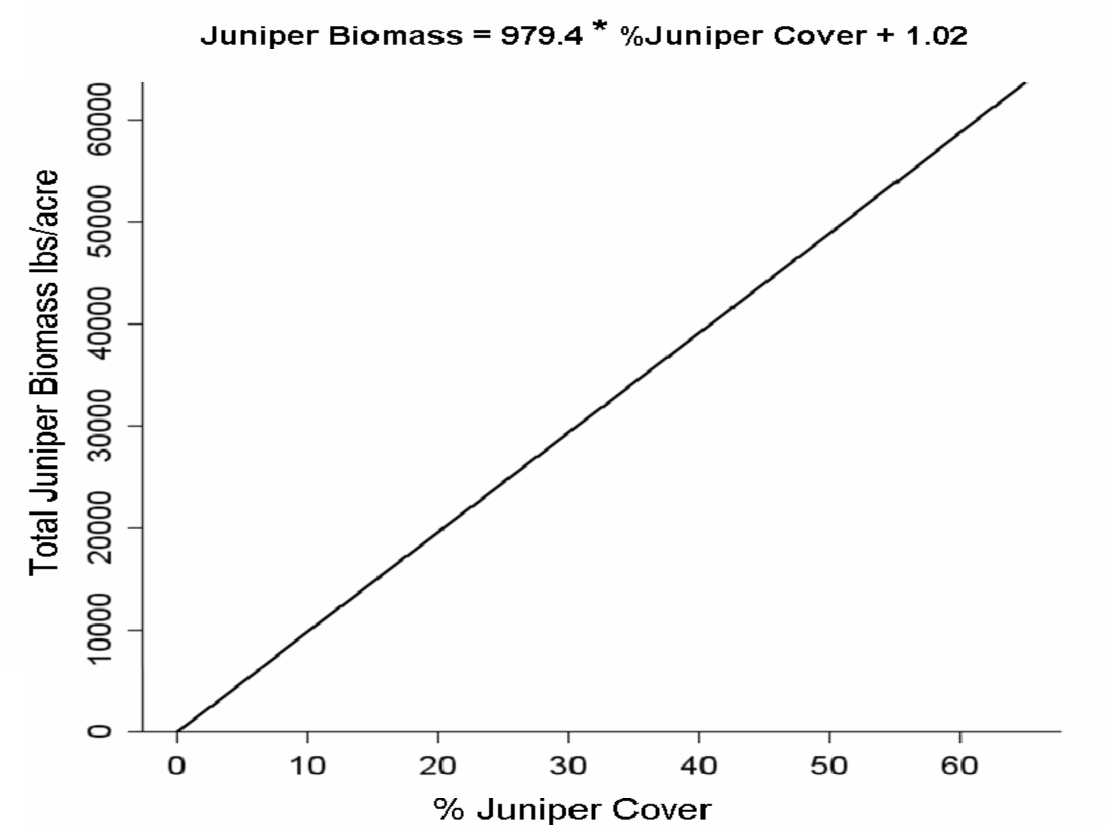


Figure 2. The relationship between percent cover and biomass of western juniper across the three study locations (n=99 trees).

Table 2. The relationships between canopy cover and the above ground biomass of the four fuel classes and the total biomass of western juniper at a landscape scale. Dry weight is expressed in lbs (y). Juniper canopy area isexpressed in acres (x).

Fuel Class Category	Equation	R^2	*P*
1-hr	y = 4.43 +19870.5x	0.55	<0.0001
10-hr	y = 9.83 + 12385.5x	0.70	<0.0001
100-hr	y = -30.51 + 29436.53x	0.78	<0.0001
1000-hr	y = -19.3 + 36322.23x	0.75	<0.0001
Total	y = 1.5 + 98011.7x	0.81	<0.0001

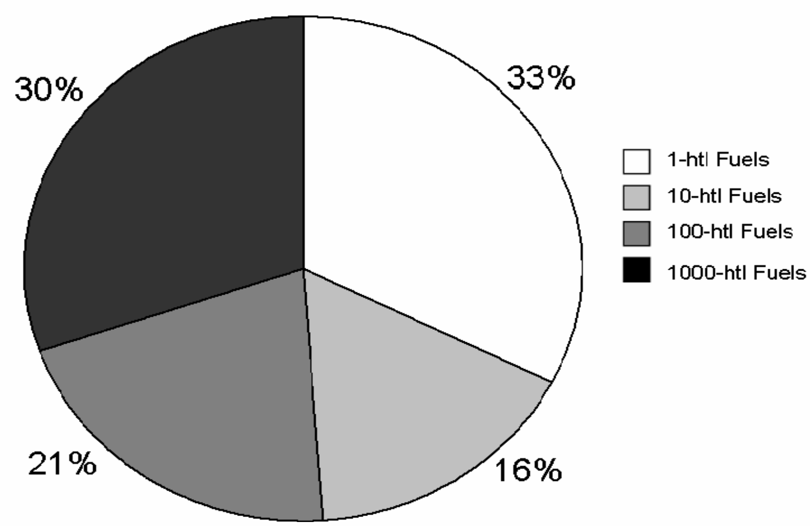


Figure 3. The average proportion of the different fuel classes per individual western juniper.

REFERENCES

Birdsey, R.A., and H.T. Schreuder. 1992. An overview of forest inventory and analysis estimation procedures in the eastern United States - with an emphasis on the components of change. USDA Forest Service General Technical Report INT-GTR-129.

Brown, J.K., R.F. Oberheu, and C.M. Johnston. 1982. Handbook for inventorying surface fuels and biomass in the interior west. USDA Forest Service General Technical Report. INT-129.

Burkhardt, J.W., and E.W. Tisdale. 1976. Causes of juniper invasion in southwestern Idaho. Ecology 57:472-484.

Gedney, D.R., D.. Azuma, C.L. Bolsinger, and N. McKay. 1999. Western juniper in eastern Oregon. U.S. Forest Service General Technical Report NW-GTR-464.

Gholz, H.L. 1980. Structure and productivity of *Juniperus occidentalis* in central Oregon. American Midland Naturalist 2:251-261.

Johnson, D.D., and R.F. Miller. 2006. Structure and development of expanding western juniper woodlands as influenced by two topographic variables. Forest Ecology and Management 229:7-15.

Miller, R.F., J.D. Bates, T.J. Svejcar, F.B. Pierson, and L.E. Eddleman. 2005. Biology, Ecology, and management of Western Juniper (*Juniperus occidentalis*). Oregon Sate University Agricultural Experiment Station Technical Bulletin 152, Corvallis.

Miller, R.F., and J.A. Rose. 1995. Historic expansion of *Juniperus occidentalis* (western juniper) in southeastern Oregon. Great Basin Naturalist 55:37-45.

Miller, R.F., and J.A. Rose. 1999. Fire history and western juniper encroachment in sagebrush steppe. Journal of Range Management 52:550-559.