Ecology and Hydrology of Western Juniper

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TABLE OF CONTENTS

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Watershed Response to Western Juniper Control: History of Camp Creek Paired Watershed Study

Tim Deboodt and Michael P. Fisher

INTRODUCTION

According to U.S. Forest Service publication PNW-RB-249, *The Western Juniper Resource of Eastern Oregon*, western juniper's dominance on eastern Oregon rangelands has increased significantly since 1934 (Azuma et al. 2005). Azuma et al., (2005) estimated that land occupied by western juniper has increased from 1.5 million to 6.5 million acres since the 1930s. Implications of this increase include loss of native, herbaceous plant communities and the bird and animal species that rely on them, increased soil loss, and reduced water infiltration. Based on water use models for individual trees, the U.S. Forest Service estimated 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 in a 13 inch precipitation zone (Gedney et al. 1999).

Soil erosion rates from sites with higher than the natural range of variability for western juniper cover were an order of magnitude greater than similar sites that are within the natural range of cover (Buckhouse and Gaither 1982). Research has shown that junipers did increase soil loss rates due to the associated decline in herbaceous ground cover and elevated surface runoff (Buckhouse and Gaither 1982; Bates et al. 2005). The juniper canopy intercepts rain and snow, keeping it from reaching the ground thus making it unavailable for plant growth, stream flow, or groundwater recharge; and they consume large amounts of soil moisture. Previous monitoring of juniper control projects has focused on changes in vegetative composition and productivity (Bates et al. 2005). These studies have usually not monitored the hydrologic impacts of western juniper control.

This project was unique in that it involved a paired study approach to monitoring changes in a watershed's water budget following western juniper control. The value of a paired watershed study is that the impacts of the treatment can be compared to the untreated watershed. This study was unique in that it is the only long-term study of western juniper ecosystems of its kind in the Pacific Northwest. Because of the time and expense in monitoring treatment responses at the watershed level, such watershed comparison studies are rarely undertaken. Similar studies in different ecological and climatic zones have been conducted in Wyoming, Utah, Colorado and Arizona (Sturges 1994, McCarthy and Dobrowolski 1999, Bosch and Hewlett 1982) but no paired watershed studies have been implemented in western juniper ecosystems.

PROJECT DESCRIPTION

The Camp Creek Paired Watershed study was initiated in 1993 to study the effects of western juniper removal on sediment yield, water yield and vegetative conversion (Fisher 2004). Two watersheds, Mays and Jensen, were identified in the Camp Creek drainage, a tributary of the Crooked River, Deschutes River Basin. Mays and Jensen were named after the original homesteaders in the area. The study area was created with the primary intent of calibrating and monitoring two watersheds for a period of time for the purpose of understanding the comparative relationships of vegetation, geomorphic and hydrologic parameters prior to treatment. Pretreatment monitoring and analysis occurred from 1994 through 2004 (Fisher 2004).

PROJECT HISTORY

This project was initiated as part of a tri state effort funded by an Environmental Protection Agency (EPA) grant to look at arid land hydrologic issues in Oregon, Nevada, and California. In 1993, the Mays and Jensen watersheds were selected and monitoring of various attributes commenced. Each watershed was delineated on the upper bounds by its ridge-tops and the lower ends designated by the placement of a channel flume. Mays watershed is approximately 280 acres and Jensen is approximately 260 acres.

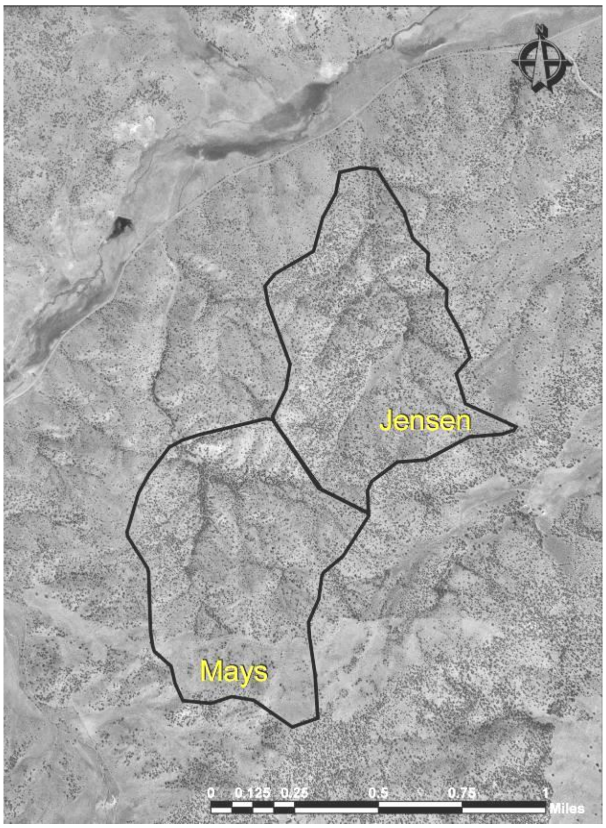

Figure 1. Aerial photograph of project area, 2004

The watersheds are located on the west branch of Camp Creek. Fourteen to twenty-five percent of each watershed was under private ownership and the remaining part study area is public land under the management of Prineville District, Bureau of Land Management (BLM) (Table 1).

Table 1. Federal versus private ownership of the study area

Mays	75% BLM	25% Private
Jensen	86% BLM	14% Private

The general orientation of both watersheds is to the north. Livestock grazing occurred in the project area and was administered by BLM under the guidance of the Brothers-La Pine Resource Management Plan (RMP)(Figure 2).

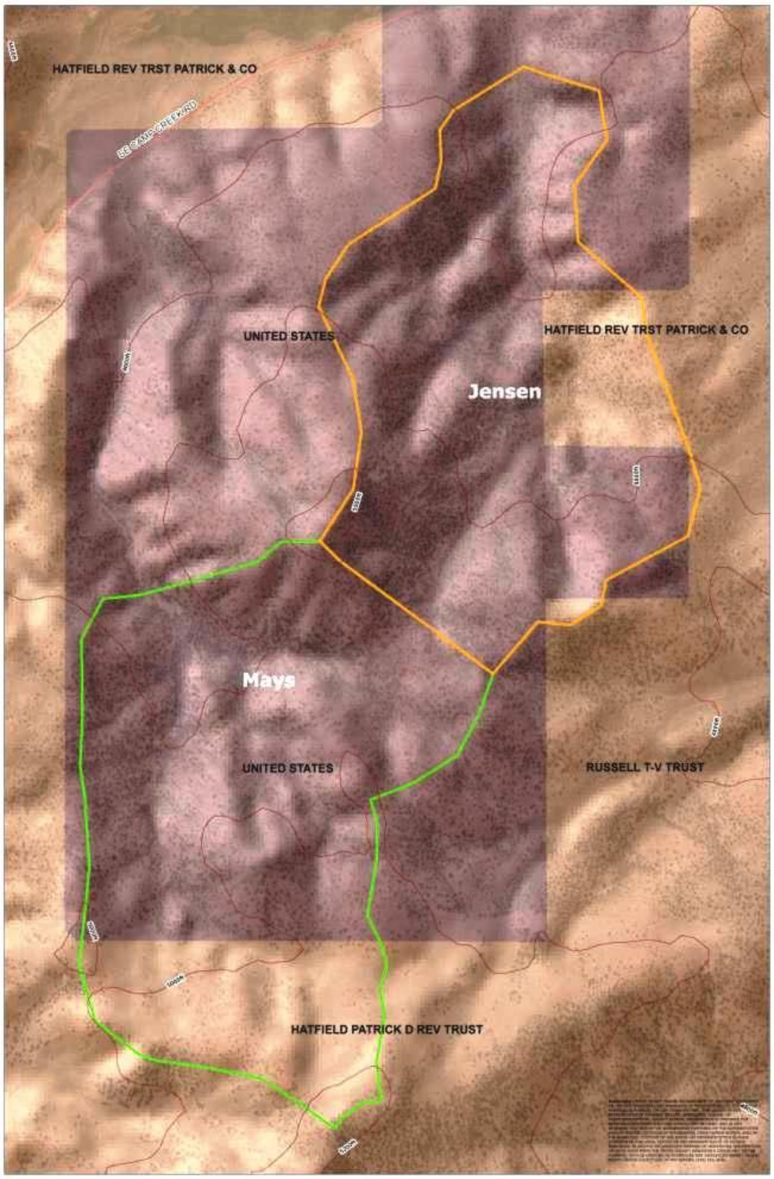

Figure 2. Ownership of project area.

Pretreatment monitoring of the area occurred from 1993 to 2004. Monitoring parameters were vegetation composition, hillslope soil movement, and channel morphology and flow. Precipitation was collected onsite and weather data was compared with Barnes Station, a USGS weather station located approximately 10 miles east of the project area. Fisher (2004) analyzed the comparative similarities and differences between the two watersheds. These comparisons provided the basis for analyzing post-treatment effects. As a result of pre-treatment analysis, additional parameters were added to the data collection protocol; the monitoring of relative soil moisture, spring flow, and the sub-surface distance to ground water were added in 2003. Mays watershed was selected as the treatment watershed and in 2005, following 12 years of pretreatment monitoring in both watersheds, all post-European aged juniper (juniper < 140 years of age) were cut in Mays. By June of 2006, all trees had been cut and bole wood from approximately 11 acres was removed. Bole wood removal from the rest of the watershed was completed over time.

METHODS

The project site is located approximately 65 miles southeast of Prineville, Oregon. Mays and Jensen watersheds are tributaries to the west branch Camp Creek, a tributary of the Crooked River, a sub-basin within the Deschutes River Basin. The study area is located within Section 32 and 33, T18S, R20E and Section 5, T19S, R20E Willamette meridian. The area is located at the southern end of the John Day Ecological Province (Anderson et al. 1998). The project site varied in elevation from 4500 to 5000 feet and the 30-year annual precipitation (1971 – 2000) at Barnes Station was 13 inches. Sixty percent of the precipitation occurred from October through March with only 25 percent falling during the growing season of April – June (Oregon Climate Service). Temperatures range from mean daily maximum of 86 degrees Fahrenheit in August to mean minimum low of 19 degrees F in February, with extremes recorded of 102 degrees F and -30 degrees F.

OBJECTIVES

The purpose of this study was to quantify the impact – on a watershed scale – of juniper control on the availability of water (quantity and timing) for beneficial uses (water quality, fisheries, irrigation, recreation, etc.) as defined by Oregon State Statute. The study involved a paired watershed approach for evaluating changes in a system's water budget following western juniper control. Water budget was measured in terms of inputs (precipitation) and outputs (soil moisture, runoff, groundwater recharge and evapotranspiration). Watershed impacts included the water budget impacts plus changes in vegetation composition and cover, and erosion rates.

Monitoring water yield following juniper control had previously not been done in the western juniper vegetation type. The value of a paired watershed study was that the impacts of the treatment could be compared to the untreated watershed. The treatment was to control western juniper in one of the watersheds. Juniper control included the cutting of all post-European-aged junipers (juniper less than 140 years of age).

Study objectives were the following:

- Measure hydrologic changes following juniper removal on a watershed scale;
- Evaluate changes in timing, duration and quantity of water expressed in channel flow, spring output, groundwater and soil moisture;
- Calculate changes in hillslope and channel morphology following juniper control;
- Quantify changes in plant community composition following juniper control.

In addition to changes in site condition, the wood products industry began to develop an interest and commercial market in western juniper. As part of the treatment activities, a harvest system was evaluated for costs of extracting juniper boles for use in log homes, dimensional wood, and fence post/fire wood. Analysis of harvest information provided land managers with information that can be used in determining opportunities for adding value and benefits to juniper control projects (Dodson and Deboodt 2007).

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Overstory-Understory Vegetation and Soil Moisture Relationships: Camp Creek Paired Watershed Study

Carlos Ochoa, Phil Caruso, Grace Ray, Tim Deboodt

SUMMARY

The effects of western juniper (*Juniperus occidentalis*) control on vegetation and topsoil water interactions were studied at the watershed-scale. Seasonal differences in topsoil water content, as affected by vegetation structure and soil texture, were determined for a pair of previously treated and untreated watersheds. A watershed-scale characterization of vegetation canopy cover and soil texture was completed to determine the driving factors influencing soil water content fluctuations throughout dry and wet seasons for one year (2014-2015). Total canopy cover, and more specifically functional group cover, was the dominant variable affecting soil water content over time. Increases in perennial grass cover were positively correlated with changes in soil water content during the wettest months. Increases in juniper cover were negatively correlated with soil water content. Soil particle analysis of samples collected from the top five inches profile fell mostly under sandy loam textural class. A few areas within each watershed showed relatively higher clay content. A geospatial analysis of soil water content and clay content showed corresponding areas of high clay and high soil water content across watersheds. Maps derived from the geospatial analysis illustrate the progression from dry to wet season, as well as the influence of topographical features on soil water content.

INTRODUCTION

The relationship between soil water content and vegetation are highly impacted by the ongoing shift from shrub steppe and grassland to woodland-dominated landscapes (Breshears et al. 1997; Gifford and Shaw 1973; USDA 1985), which has the potential of modifying the ecological and hydrological balance of these water-limited regions (Huxman 2005; Owens 2006; Yager and Smeins 1999). In many areas of the western United States, the significant expansion of juniper (*Juniperus* spp.) observed over the last two centuries is disrupting important ecological and hydrological functions. Juniper encroachment can limit the growth of shrubs, grasses, and forbs, by outcompeting them for light, soil moisture, and soil nutrients (Gottfried and Pieper, 2000; Vaitkus and Eddleman 1987), reduce biodiversity (Tausch and West 1995; Miller et al. 2000; Bates et al. 2005), modify hydrologic processes (Mollnau et al. 2014; Zou et al. 2013; Petersen and Stringham 2008; Wilcox 1994), and alter soil nutrient cycling (Bates et al. 2002).

Many of these studies, which have been conducted at the plot-scale, heightened the need for evaluating juniper encroachment effects on vegetation and hydrological processes at a larger spatial scale. Wilcox and Thurow (2006) discussed the emerging issues related to juniper encroachment and the need to complete landscape-scale studies detailing ecosystem wide feedbacks that react to encroachment. Our study aimed to enhance base knowledge of the effects that western juniper encroachment has on vegetation and soil water dynamics at the watershed scale. The main objective was to determine vegetation and soil water dynamics on two adjacent watersheds, one treated (~ 90% of the western juniper removed) and one untreated.

METHODS

The study area covers approximately 500 acres and it encompasses two adjacent (one treated and one untreated) watersheds with similar dimensions. The average percent slope for each watershed was measured at ~ 25% (Fisher 2004). The distributions of aspects are also similar across both watersheds at ~ 35% north-facing slopes and ~ 25% west-facing slopes. Vegetation and topsoil moisture dynamics were evaluated across these two watersheds. A total of 289 ten-meter transects were installed across watersheds to collect soil moisture, vegetation, and soil data (Figure 1). Transect locations were distributed through the two watersheds (Treated, n = 143; Untreated, n = 146) to provide a fair representation of aspect and elevation. Overstory and understory vegetation cover data were recorded by species functional groups, which were categorized as forb, annual grass, perennial grass, shrub, and tree. Vegetation data was collected every one meter (3.3 feet) in each transect and was used to estimate cover for each functional group, bare ground, and litter cover, and for estimating total canopy cover by all vegetation species combined in each watershed. Soil samples for determining topsoil texture and measurements of soil moisture were collected every two meters (6.6 feet) in each transect. A portable probe was used to collect 1,445 soil moisture measurements of the top five inches soil profile during each of five selected months between August 2014 and May 2015. The hydrometer method was used to determine soil texture based on particle size distribution values obtained from each soil sample collected.

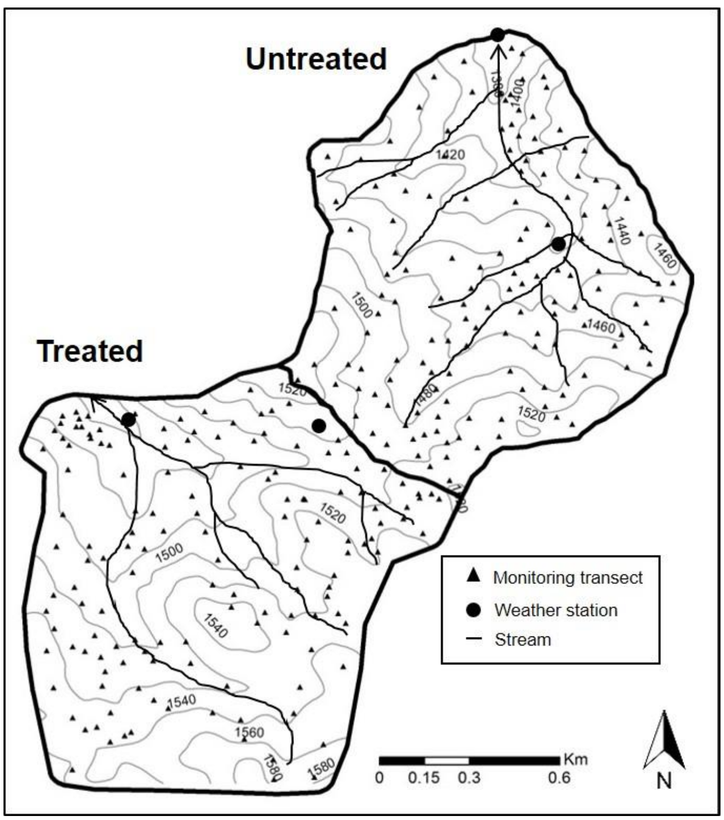

Figure 1. Map of the study area illustrating monitoring transect distribution throughout the watersheds.

Juniper canopy interception and soil water relationships were evaluated in the untreated watershed starting in October 2005. One soil monitoring station was installed at the valley bottom and at a mid-hillslope elevation location within the watershed. Each station consisted of two vertical networks of three soil moisture sensors collocated at different soil depths (8, 20, and 32 inches), and installed in tree undercanopy and intercanopy locations. At the valley site, ten non-recording rain gauges were installed at undercanopy (n =4), drip line (n = 3), and intercanopy (n = 3) locations. Juniper canopy cover above each rain gauge was determined using a convex spherical densiometer.

RESULTS

Canopy Cover

Litter, bare ground, and vegetation canopy cover were compared across watersheds. In general, greater litter cover and less bare ground were observed in the treated watershed when compared to the untreated. Most vegetation functional groups (shrub, perennial grass, and annual grass) showed higher canopy values in the treated watershed when compared to the untreated. Forb canopy cover was not significantly different across watersheds. As expected, juniper cover was considerably higher in the untreated watershed (36%) than in the treated (10%) watershed (Figure 2). No significant differences in total (overstory and understory combined) canopy cover were observed across watersheds.

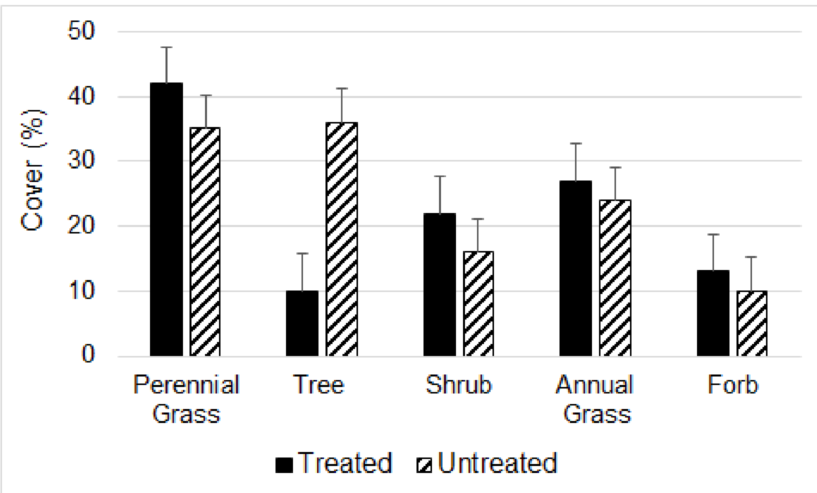

Figure 2. Mean canopy cover for each functional group across both watersheds.

Interception and Soil moisture Dynamics

Juniper canopy interception and soil water transport through the soil profile were evaluated at the valley bottom location in the untreated watershed. Juniper canopy cover ranged from 9% to 98% with mean values of 97%, 68%, and 32% for undercanopy, drip line, and intercanopy locations, respectively. Rain totaling approximately one inch (0.96) fell during the period of record October 31 to November 21, 2015. Study results show that, on average, 70% of rainfall was intercepted before reaching the rain gauges at undercanopy locations. Average canopy interception at the drip line was 29%, followed by 11% at the intercanopy. A time-lapse camera installed onsite showed snowfall was also highly intercepted by tree canopy cover during the winter. No quantification of snowfall interception amount was recorded. The effects of tree canopy interception of rain and snow precipitation during the fall and winter season were evident in the soil moisture response. Figure 3 shows soil moisture for the sensors installed at undercanopy and intercanopy locations in the valley bottom between October 2015 and May 2016. Overall, higher soil moisture content was observed in all sensors installed at the intercanopy location throughout the entire period of record. At the intercanopy location, sensors installed at 20 and 32 inches depth showed considerably lower soil moisture levels than those sensors installed at the same soil depths in the intercanopy location. Soil moisture response for the 32 inches sensor in the undercanopy location was delayed nearly six months when compared to the same sensor depth in the intercanopy.

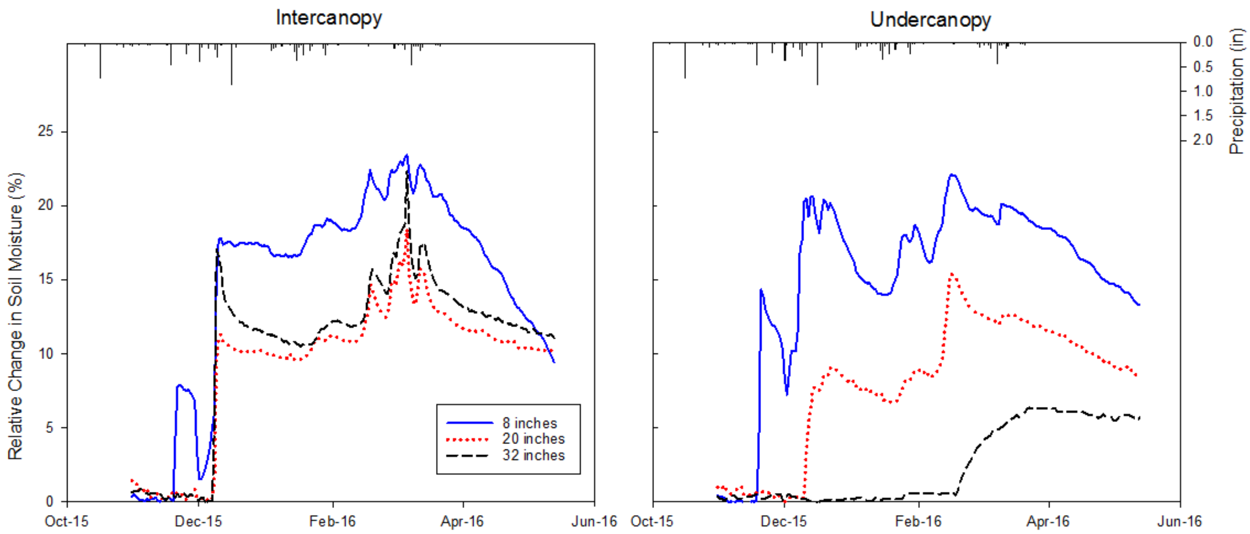

Figure 3. Soil moisture variability for sensors installed at different soil depths in intercanopy and undercanopy locations.

A progressive change in soil moisture corresponding to the transition from the dry to wet season was observed in both watersheds throughout the study period. Mean soil moisture values for each watershed were derived from field data collected during the data collection months (July, November, January, March, and May). In three (July, January, and May) out of the five months, slightly higher (< 3%) mean soil moisture values were observed in the treated watershed (Table 1).

Table 1. Integrated soil moisture values for the treated and untreated watersheds.

	Treated Watershed			Untreated Watershed		
	Soil moisture (%)			Soil moisture (%)		
Month/Year	Min	Max	Mean*	Min	Max	Mean*
July 2014	2	17	8.2 a	3	12	7.08 b
November 2014	4	16	9.9 a	5	15	10.0 a
January 2015	10	40	23.7 a	9	37	20.9 b
March 2015	11	37	25.6 a	14	40	27.2 b
May 2015	17	42	28.4 a	12	41	25.7 b

* By month mean values with the same letter are not significantly different across watersheds ($P \leq 0.05$).

In general, soil moisture results showed no significant differences ($P \leq 0.05$) by aspect across watersheds for each measurement period. Using soil moisture data collected and interpolation data techniques, we developed contour maps illustrating topsoil water distribution across both watersheds in each measured month. Figures 4 and 5 show the progressive change in soil moisture from the dry to the wet seasons across both watersheds. During the dry season months (July and November), greater soil moisture values were obtained in higher elevation areas in each watershed (Figure 4). As topsoil conditions got wetter throughout winter and spring, greater soil moisture values were observed at the bottom of the watersheds, near the stream channels (Figure 5).

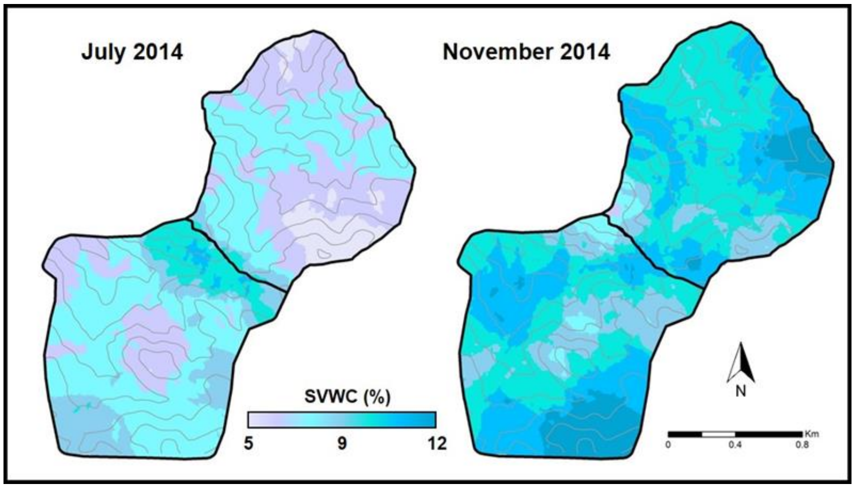

Figure 4. Map of the research area illustrating topsoil moisture, expressed as percent soil volumetric water content (SVWC), distribution throughout the two driest monitored months (July and November 2014).

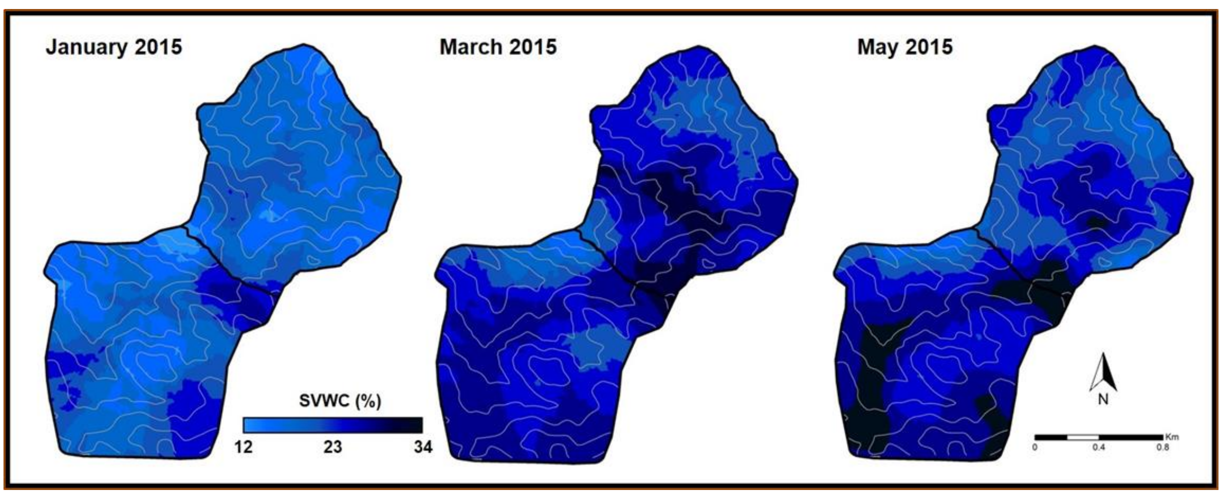

Figure 5. Map of the research area illustrating topsoil moisture, expressed as percent soil volumetric water content (SVWC), distribution throughout the wettest monitored months (January, March, and May 2015).

DISCUSSION

Results from this watershed-scale study indicate that overall, dense juniper cover can result in lower available soil moisture, particularly under tree canopy. Our findings indicate that canopy cover plays an important role in soil moisture distribution across the landscape. The progressive changes in soil moisture content observed across watersheds during the transition from the dry to the wet season can be affected by degree and type of vegetation cover. Results showed that perennial grass cover was positively correlated with changes in soil moisture, whereas juniper cover showed a negative correlation with soil moisture content. Dense tree canopy cover commonly observed in Phase III juniper stands, similar to our untreated watershed, can intercept significant amounts of precipitation therefore limiting the amount of water reaching the ground. This can be more acute during rainfall events when most of the precipitation intercepted can be lost through direct evaporation from the tree canopy. Study results provide valuable information towards understanding ecological and hydrological relationships in western juniper dominated landscapes.

MANAGEMENT IMPLICATIONS

Results from this study can provide useful information to land managers for planning of juniper removal efforts aimed to improve rangeland conditions. The effects of juniper removal may not be self-evident but it will certainly result in a redistribution of water budget components due to the lack of tree canopy interception. In turn, this can potentially influence vegetation and water distribution within and outside of the watershed. Watershed-scale analyses of ecological and hydrological interactions ought to be considered when developing land management projects aimed to maximize ecosystem services (e.g., water and forage provisioning, wildlife habitat). It is important for sound science to be tied with management objectives and desired outcomes to develop best management practices for juniper control.

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Upland-Valley Hydrologic Connectivity: Camp Creek Paired Watershed Study

Carlos Ochoa, Phil Caruso, Grace Ray, Tim Deboodt

SUMMARY

Surface water and groundwater relationships in treated (juniper removed) and untreated watersheds and their connecting riparian valley were studied. Study results show relatively rapid water transport through the soil profile and into the shallow aquifer in both watersheds. This is particularly true during the winter precipitation season. Summer precipitation events resulted in soil moisture response across the top 32-inch soil profile but did not have an effect in shallow groundwater. A longer subsurface flow residence time was found in the treated watershed when compared to the untreated. Similarly, greater springflow and runoff rates were observed in the treated watershed. Study results indicate there are temporary hydrologic connections through the shallow groundwater systems between upland watersheds and valley locations during the winter precipitation season. An isotope trace analysis showed a similar isotopic signature for upland and valley well locations, indicating there are temporary hydrologic connections through the shallow groundwater system.

INTRODUCTION

Water provisioning is the ecosystem service that most directly links human population growth and rangeland ecosystems (Havstad et al. 2007). The freshwater ecosystem service is intrinsically related to other supporting and regulating services such as soil development, water regulation, and climate regulation (MEA 2005). It is increasingly recognized that comprehensive resource management requires integration of surface water and groundwater components and that juniper expansion effects on groundwater recharge must be better understood. Hydrologic connectivity, that is, surface water and groundwater flow dynamics throughout the watershed, may be an important determinant of ecosystem resilience. Hydrologic connectivity is the most important characteristic related to short- versus long-term water management, and often it is poorly understood or characterized. The connections between upland water sources, groundwater, and downstream valleys influence the amount of water available to multiple natural processes that drive many ecosystem services (e.g., forage provisioning, wildlife habitat, recreation, etc.). Several studies have reported the temporally variable hydrologic connectivity between uplands and valleys (Detty and McGuire 2010; Jencso et al. 2009). Studies have shown there are direct connections between vegetation, hydrology, and other physical attributes such as topography and geology (Albertson and Kiely, 2001; Emanuel et al. 2014). Composition and structure of vegetation are important features that affect hydrology, nutrient and energy cycles, ecological services and disturbance regimes (Miller et al. 2013). Vegetation depends on water provisioning, but at the same is responsible for producing and maintaining the quality of this ecosystem service (MEA 2005). Most studies related to hydrologic connectivity have been done in more mesic environments. The ecologic and hydrologic linkages between upland water sources and downstream valleys in arid and semiarid regions are virtually unstudied. There is a need for more and better information regarding landscape-scale processes and land management decisions in semiarid, juniper-dominated, woodlands (Miller et al. 2005; Wilcox et al. 2006). The main objectives of this study were to 1) assess surface water and groundwater interactions in treated and untreated watersheds; and 2) characterize hydrologic connectivity between the upland watersheds and the downstream valley.

METHODS

The study area covers approximately 1000 acres and includes one treated watershed (~ 90% juniper removal), one untreated watershed, and a riparian valley, where both watersheds drain into. The wet season in the study area occurs between September and April, with the majority of the precipitation occurring as snowfall. Beginning in 2003, the study site was instrumented to record weather, streamflow, soil moisture, and groundwater level fluctuations data (see Deboodt 2008). A weather station and a flume type H were installed at the botoom of each watershed. A total of eight soil moisture stations with vertical nests of soil moisture sensors installed at 8 inches, 20 inches, and 32 inches depth are located in the riparian valley and at upland and bottom locations in both watersheds. Four of these stations were installed in 2003 and the other four have been recently installed (2015-2016). Transects of six wells installed perpendiculary to the stream were installed at the outlet of each watershed in 2003. In order to better understand upland-valley hydrologic connections, we have recently installed a cluster of three monitoring wells in the valley downstream of these watersheds. All wells (new and old) have been equipped with stand-alone water level loggers. Also, we have added a snow gauge and a rain gauge at the watershed divide. Field estimates of springflow rate have been obtained at selected dates since 2003 using a one gallon container and a stop watch.

In the spring of 2015, we conducted an isotope trace anlaysis to determine potential similarities, or discrepancies, between different water sources across the study site. Samples were collected in both upland watersheds and in the riparian valley. Upland sources included precipitation collected from the rain gauge located at the watershed divide, one spring source and two wells in the treated watershed, and one spring source and three wells from the juniper dominated watershed. Also, samples from two wells in the riparian valley were collected.

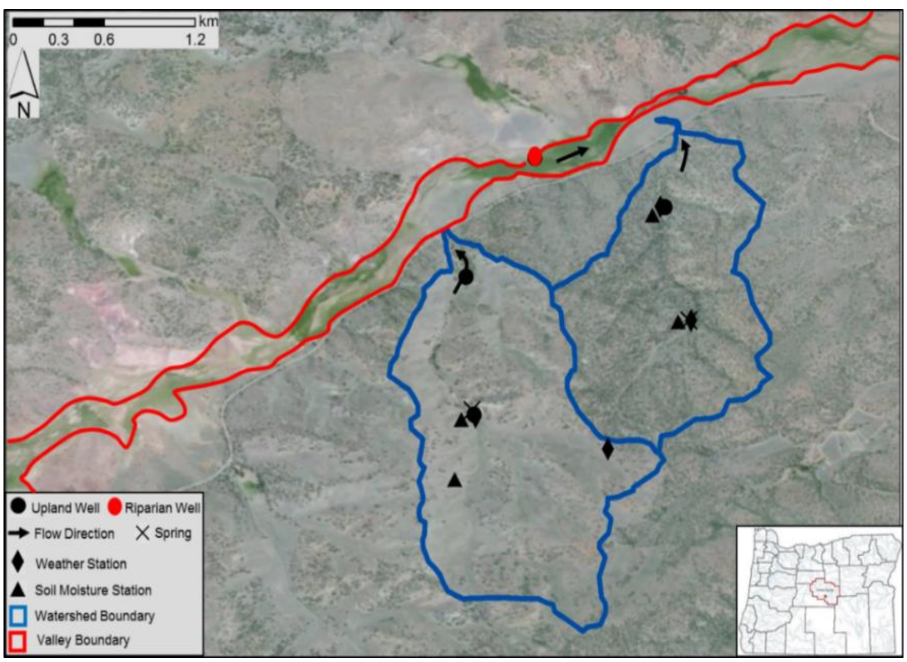

Figure 1. Study area illustrating instrumentation installed.

RESULTS

In general, higher runoff and springflow flow rates, and an increase in shallow groundwater residence time were observed in the treated watershed, when compared to its adjacent, heavily encroached, watershed. Figure 2 shows the seasonal recharge of the soil profile and shallow aquifer response to precipitation inputs during June 2014 through June 2015.

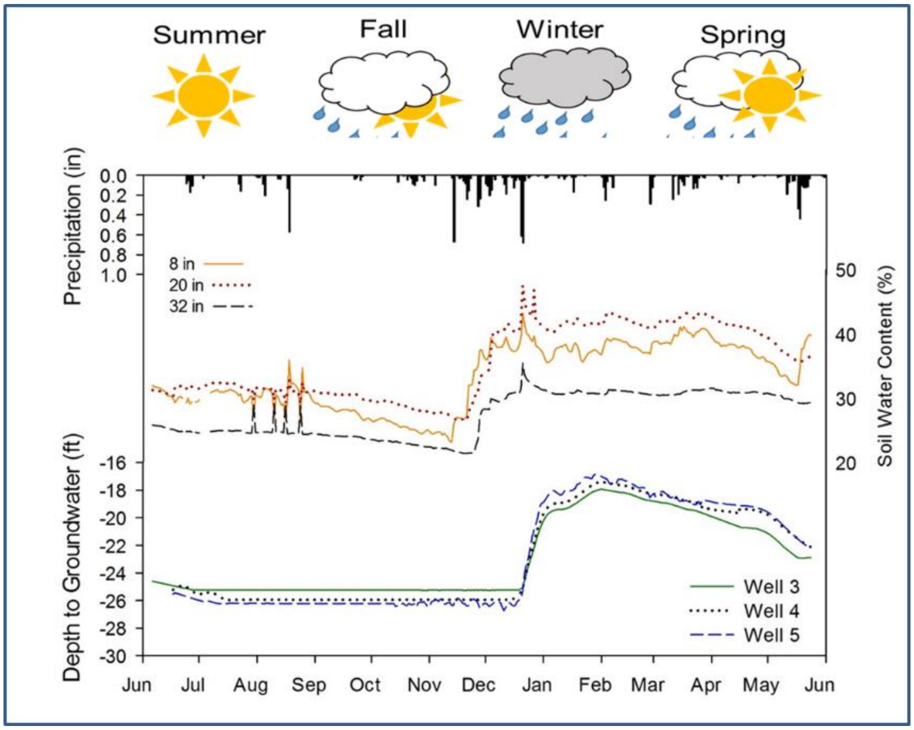

Figure 2. Soil moisture and groundwater level response to precipitation inputs in the untreated watershed (2014-2015).

Results provide valuable information regarding precipitation effects on soil moisture response at shallow (8 inches) and deeper (20 and 32 inches) soil depths. A relatively rapid rise and decline in soil moisture level was observed during specific isolated storms in the summer season. After the transient response to individual rainfall events during the summer, soil moisture steadily declined until its lowest level at each sensor depth in mid- November through early December. As more precipitation occurred during the fall and winter, soil moisture levels in the soil profile began gradually increasing. The sensor installed at 8 inches depth responded first, followed by the 20 inches depth, then by the sensor installed at the deepest 32 inches. Once the top 32-inch soil moisture reached near saturation, a sharp rise in shallow groundwater level was observed in wells installed in the upper watersheds (Figure 2). In general, both watershed showed similar dynamics of precipitation water movement through the soil profile and into the shallow aquifer. Shallow groundwater levels in the untreated watershed were higher than in the treated watershed but also declined faster.

The longer residence time in shallow groundwater observed in the treated watershed was also reflected in greater springflow rates and longer periods of flow. Springflow levels in the treated watershed have been consistently higher than in the untreated watershed even before juniper removal. However, after the juniper removal effort that took place between 2005 and 2006 a substantial difference in the number of springflow days in both watersheds has been evident (Figure 3). This increase in the number of springflow days has been previously documented by Deboodt (2008). A peak flow rate of 50 gallons per minute has been documented several times throughout the study period since 2006.

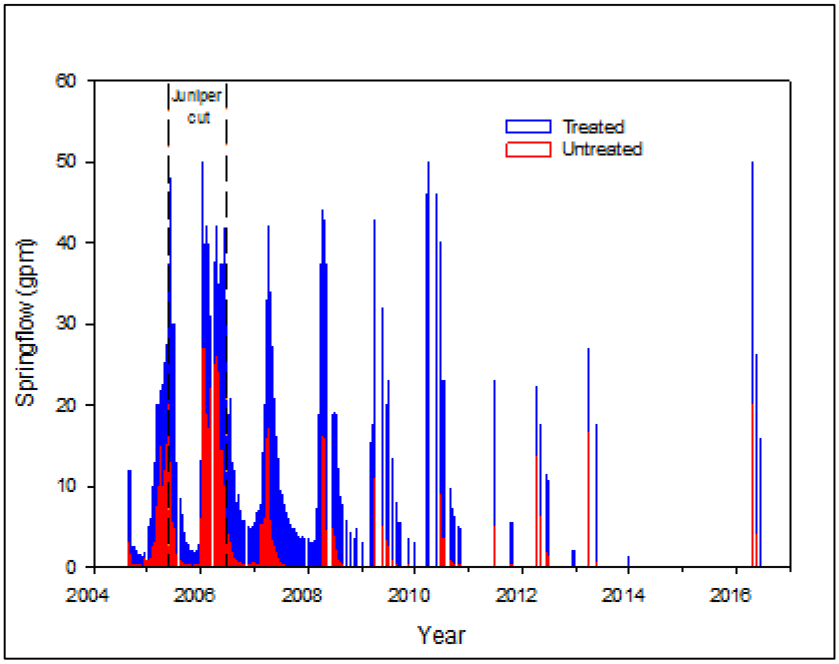

Figure 3. Long-term manual measurements of springflow in both watersheds.

The connections between surface water and groundwater were more apparent in the treated watershed than in the untreated. Figure 4 shows surface runoff and springflow values for both watersheds during 2016. Peak surface runoff in the treated watershed rose 116 gpm, which was substantially higher than the peak runoff value of 4 gpm observed in the untreated watershed. Surface runoff data and flume pre-calibrated equations were used to calculate total water yield for the treated (23 acre-feet) and untreated (0.4 acre-feet) watersheds.

Springflow maximum rates of 50 gpm for the treated watershed, and 20 gpm for the untreated watershed were obtained in 19 April. After that, springflow rates in both watersheds steadily declined to 16 gpm (treated) and 2 gpm (untreated) in 14 June.

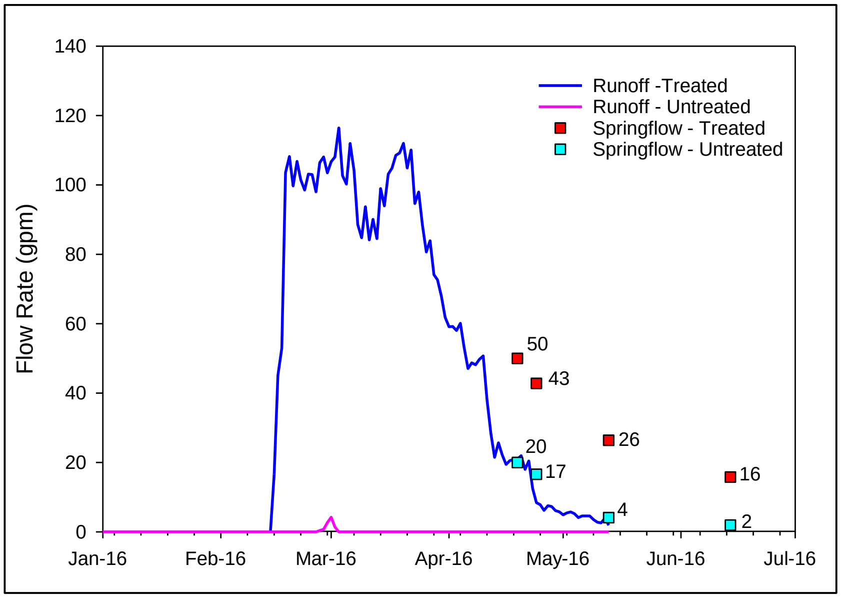

Figure 4. Surface runoff and springflow values of treated and untreated watersheds during 2016.

Springflow flow extended several weeks past spring surface runoff, particularly in the treated watershed (see Figure 4). This increased residence time through the subsurface flow system is critical to maintain hydrologic connections within and out of the watersheds with the downstream valley. Upland-valley shallow groundwater connections are more evident during the wet season. Figure 5 illustrates a sharp water level rise observed in an upland well location early in the winter season, along with the late response observed in a well located in the riparian valley. The well in the valley shows a more gradual rise observed throughout the season until early March when the level abruptly increased, likely due to stream seepage contributions from a nearby stream fed by upland subsurface water flow contributions. This shallow groundwater level rise in the riparian valley well during the winter season showed a 4 to 6 week delay when compared to the upland well location (Figure 5). Results from the isotope analysis showed close similarity in values across all groundwater sources, which further points to the connective nature of the upland water sources and the downstream valley at the study site.

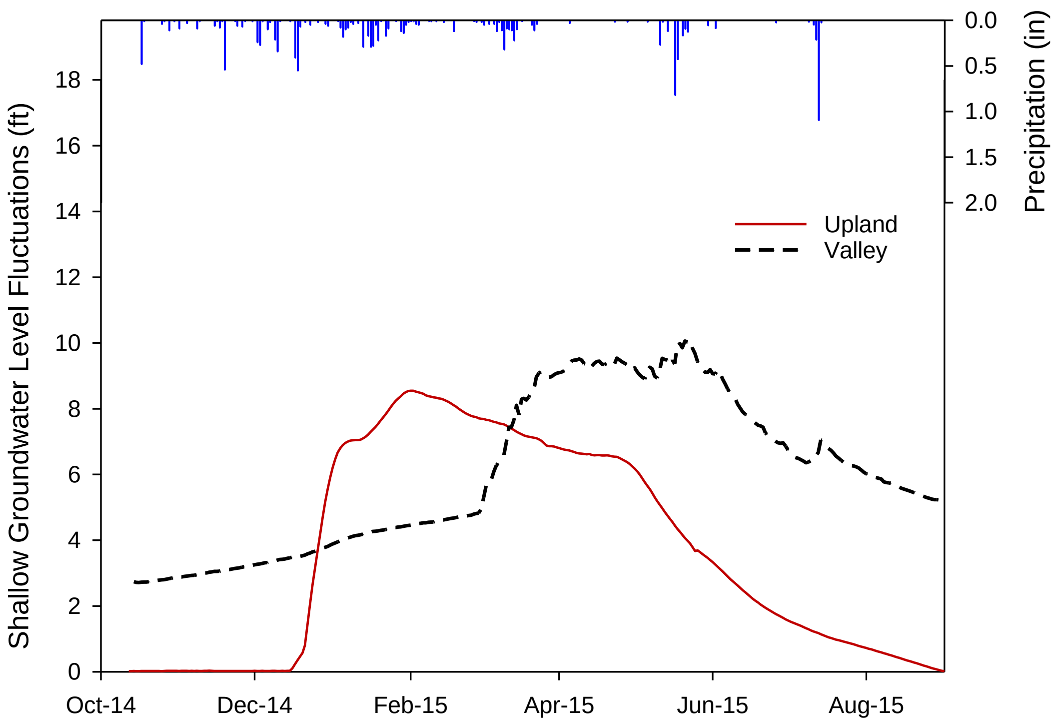

Figure 5. Seasonal shallow aquifer response to precipitation inputs observed in upland and valley wells (2014-2015).

DISCUSSION

One of the objectives of this study was to characterize surface water and groundwater relationships in a phase III juniper watershed and its adjacent watershed where 90% of the juniper was removed in 2006. Results show greater runoff and springflow rates were observed in the watershed where juniper was removed ten years ago. Even though springflow rates have always been higher in the treated watershed, the post-treatment increase in the number of days with springflow (Deboodt 2008) suggests this may be due to the effects of juniper removal. As discussed in the manuscript above, the observed high levels of juniper canopy interception may have played a role in preventing fair amounts of precipitation from reaching the ground in the untreated watershed. The effects of clearing overstory vegetation are mostly noticed at the bottom of the watersheds where surface and subsurface flows tend to concentrate. Long-term seasonal increases in shallow groundwater residence time and an upward trend in soil moisture in the lower monitoring station following juniper removal at the treated watershed have been previously reported by Ray (2014). This is consistent with the higher runoff and springflow rates observed in the treated watershed in this study. A second objective of this study was to characterize the hydrologic connections between upland water sources and the downstream valley. Shallow groundwater level response observed in upland and valley monitoring wells, associated with the observed surface and subsurface flow dynamics, and the results from the isotope trace analysis, all indicate there are temporary hydrologic connections between upland and valley locations during the winter precipitation season.

MANAGEMENT IMPLICATIONS

Study findings provide a better understanding of surface water and groundwater interactions in western juniper dominated watersheds of central Oregon. Study results provide an improved understanding of the transient hydrologic connections within treated and untreated watersheds and between upland water sources and downstream valleys. A comprehensive understanding of the mechanisms of water distribution within and out of the watershed is important when designing resource management projects aimed to improve the overall watershed function. It is important that best management practices for juniper control be based on solid scientific understanding of the ecological and hydrological interactions occurring in these dryland ecosystems.

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Estimating Juniper Cover from NAIP imagery and Evaluating Relationships between Potential Cover and Environmental Variables

Dustin D. Johnson, Kirk W. Davies, and Jon D. Bates

SUMMARY

Juniper management is constrained by limited tools to estimate juniper cover and potential cover at stand closure across landscapes. We evaluated if remotely sensed imagery (NAIP) could be used to estimate juniper cover and if environmental characteristic could be used to determine potential juniper cover at stand closure. We determined that reasonably accurate estimates of western juniper cover can be obtained from NAIP imagery. We also found that environmental characteristics could explain 40% of the variability in juniper cover at stand closure.

INTRODUCTION

Control of western juniper that has encroached into sagebrush rangeland has been shown to increase understory productivity, cover, and diversity (Evans and Young 1984; Rose and Eddleman 1994; Bates et al. 1998; Bates et al. 2006). However, the effectiveness and associated cost of management actions largely depends on the structural attributes and developmental rates of juniper stands which can vary strongly across landscapes (Johnson 2005; Johnson and Miller 2006; Petersen et al. 2009). Selection of the most effective management actions and prioritization of juniper control across landscapes is critical for obtaining the most economic and ecologic benefit from limited management resources. In order for land managers to prioritize juniper woodlands for treatment and select the most effective management action, information on current and potential juniper cover is needed across variable landscapes. However, to date, there are two major constraints, 1) current forest and rangeland inventory methods are time consuming and expensive and 2) landscape estimates of potential western juniper cover at stand closure are lacking. Remotely sensed images covering large areas may represent an opportunity to monitor and inventory western juniper encroachment inexpensively relative to standard rangeland and forest inventory methods. If relationships between commonly available geospatial data layers and potential juniper cover can be determined, then estimates of potential juniper cover across landscapes may be feasible.

The purposes of this study was to determine 1) the efficacy of using widely available NAIP imagery to estimate western juniper cover, and 2) the relationship between environmental/site characteristics and juniper cover in closed western juniper stands across heterogeneous sites. We hypothesized that several recently developed indices of environmental gradients and/or basic site characteristics may have utility for estimating juniper cover at stand closure. Specifically, an integrated moisture index (IMI) (Iverson et al. 1997), a site exposure index (SEI) (Balice et al. 2000) and a heat load index (HLI) (McCune and Keon 2002) were tested in this study because of the relative ease of application to landscapes and their potential biological significance.

Methods

The 12,340 ha study area was located on Juniper Mountain in Owyhee County, Idaho between the towns of Grand View, Idaho and Jordan Valley, Oregon. Mean annual precipitation ranges from 300 mm at lower elevations increasing to > 560 mm at higher elevations and is primarily received in fall, winter and early spring. Average minimum and maximum temperatures vary from -6.6 and 3.3^{o} C in January to 13.3 and 34.5^{o} C in July, respectively. The growing season ranges from 90 to 120 days across most of the study area, but is less than 60 days at higher elevations. Soils vary from shallow rock outcrops to moderately deep gravelly, sandy, or silt loams (Harkness 1998). Predominant soil taxa are Aridisols, Entisols, Alfisols, Inceptisols, and Mollisols, which occur in combination with mesic and frigid soil temperature regimes and xeric and aridic soil moisture regimes. Cryic temperature regimes occur at higher elevations typically above the western juniper woodland belt (600 – 2100 m). The major potential plant associations across the valley slopes and bottoms are: 1) mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana* Rydb.) associated with either bluebunch wheatgrass (*Agropyron spicatum* Pursh) or Idaho fescue (*Festuca idahoensis* Elmer) on relatively deep, well-drained soils and 2) low sagebrush (*Artemisia arbuscula* ssp. *arbuscula* Nutt.) associated with bluebunch wheatgrass, Idaho fescue, or Sandberg bluegrass (*Poa sandbergii* Vasey) over restrictive layers of claypan or bedrock (Burkhardt and Tisdale 1976). These plant associations are common across the Intermountain West (Miller and Eddleman 2000; Davies et al. 2006). Sagebrush plant communities encroached by western juniper woodlands were the focus of this investigation.

A completely random design was used to compare estimates of juniper cover derived from NAIP imagery to ground measurements. Forty points were randomly selected across the 12,340 ha study area. The nearest closed juniper stand to each randomly selected point was selected for sampling. Juniper cover values at the selected plots were estimated with ground measurements and from NAIP imagery. Ground measurements were conducted in one 30 x 50 m plot at each randomly selected stand. Juniper cover was measured using the line intercept method along three 50-m transects spaced at 15-m intervals. The randomly selected, closed juniper plots were also used to determine correlations between juniper cover at stand closure and the environmental/site variables and indices. Environmental/site characteristics and indices were derived from USGS 10-m digital elevation model (United States Geographical Service 2008), except soil characteristics were derived from NRCS Soil Survey Geographic Database SSURGO map files (Natural Resource Conservation Service 2010).

Results

Juniper cover classification showed an overall accuracy of 92%, and Kappa statistic of 0.84. Juniper cover estimates from NAIP imagery and ground measurements were strongly correlated (Fig. 1). There was high agreement between the ground measurements and estimate from NAIP imagery ($R^2 = 0.74$, $P < 0.01$). Minimum juniper cover recorded was 26.8 and 24.7% using the aerial images and ground measuring methods, respectively. Maximum juniper cover recorded was 82.2 and 78.7% using the NAIP imagery and ground measuring methods, respectively. Mean difference between NAIP imagery and ground measured juniper cover was 6.6 ± 0.61% ($P < 0.01$). Minimum and maximum difference between NAIP imagery and ground measurements was 0.04% and 13.5%, respectively. However, juniper cover estimates derived from NAIP imagery compared to ground measurements were not consistently higher or lower (P = 0.79).

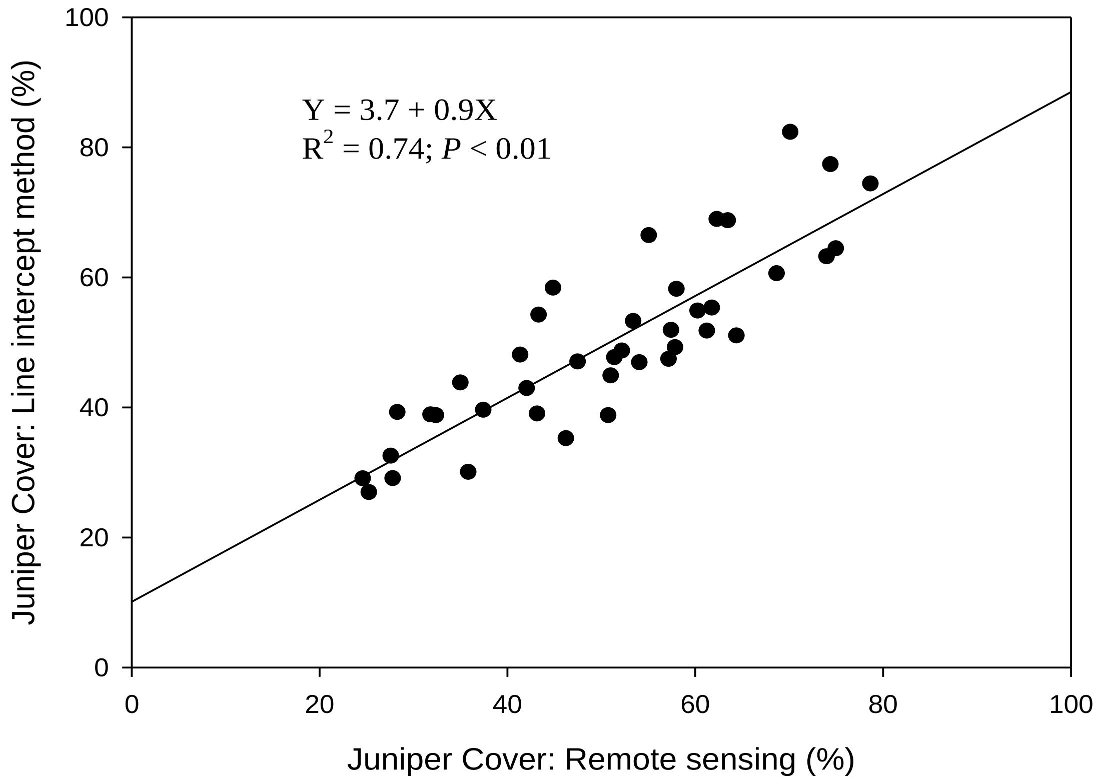

Figure 1. Correlations between ground and remote sensed estimates of western juniper cover at Juniper Mountain, Idaho.

Correlations between environmental gradient indices and juniper cover at stand closure were either not significant or only explained a limited amount of variation in juniper cover. The IMI and juniper cover in closed stands were not correlated (P = 0.68). Similarly, the HLI was not correlated with juniper cover at stand closure (P = 0.74). The juniper cover at stand closure was correlated negatively with SEI (P = 0.04). The SEI explained 10% of the variation in juniper cover at stand closure (R^2 = 0.10).

Environmental variables were correlated to juniper cover in closed woodlands. Juniper cover at stand closure correlated positively with aspect, slope, and elevation and correlated negatively with the interaction between slope and aspect. The linear regression model best describing the relationship between juniper cover at stand closure and environmental characteristics was (standard errors in parentheses below parameter estimates):

$$\text{Juniper cover} = \underset{(44.17)}{-49.62} + \underset{(6.27)}{1.69}\,(\cos(\text{aspect})) + \underset{(0.45)}{0.90}\,(\text{slope}) + \underset{(0.02)}{0.05}\,(\text{elevation}) - \underset{(0.71)}{1.32}\,(\text{slope}*\cos(\text{aspect}))$$

This equation explained 40% of the variation in juniper cover at stand closure (R^2 = 0.40; P < 0.01).

Discussion

The results of this study demonstrate that reasonably accurate estimates of western juniper cover can be obtained from NAIP imagery. This suggests that aerial images, in conjunction with feature extraction software, can be used to reliably estimate western juniper cover over large landscapes. Accurate estimates of western juniper cover are essential to prioritizing management and selecting the appropriate treatments in juniper control programs to restore sagebrush steppe plant communities (Miller et al. 2005).

Estimating juniper cover at stand closure based on selected environmental gradient indices proved to be ineffective. Neither the IMI nor HLI were correlated with juniper cover at stand closure (P = 0.68 and 0.74, respectively). In contrast, Davies et al. (2007) reported that the HLI explained some of the variation in vegetation cover in several plant functional groups in sagebrush plant communities; however, their study did not include western juniper. Similar to the correlation between SEI and juniper density reported by Johnson and Miller (2006), we found that the SEI was correlated with juniper cover (P = 0.044), but explained only 10% of its variation. Thus, environmental indices tested were limited in their usefulness at explaining variation in potential juniper cover for management purposes.

However, the correlation between environmental factors and potential juniper cover was stronger. The moderate correlation between environmental characteristics and western juniper cover at stand closure (R^2 = 0.40; P < 0.01) is similar to other attempts to correlate environmental characteristics with vegetation characteristics across landscapes in the Intermountain West (Jensen et al. 1990; Johnson and Miller 2006; Davies et al. 2007). Johnson and Miller (2006) found moderate to strong correlations between juniper (total and dominate tree) density and environmental characteristics. Petersen and Stringham (2008) found strong relationships between sagebrush structural characteristics and explanatory variables. However, they recognized that the correlations found between vegetation structure and explanatory variables in their study would probably be weaker if applied at large landscapes.

The relationship between environmental factors and juniper cover suggest that we should expect greater western juniper cover at stand closure at higher elevations, on steeper slopes, and in more northerly facing aspects (Fig. 2). These factors probably influence juniper cover by their influence on the availability of water to juniper trees. Less exposed sites would have reduced evaporation, thus more water would be available for transpiration. Similarly, Davies et al. (2007) reported that relationships between herbaceous cover and environmental characteristics were probably due to the influence of the environmental characteristics on availability of water for plant growth. Johnson and Miller (2006) also reported the influence of environmental/site characteristics on juniper stand characteristics was probably due to environmental/site characteristics' effect on soil water availability.

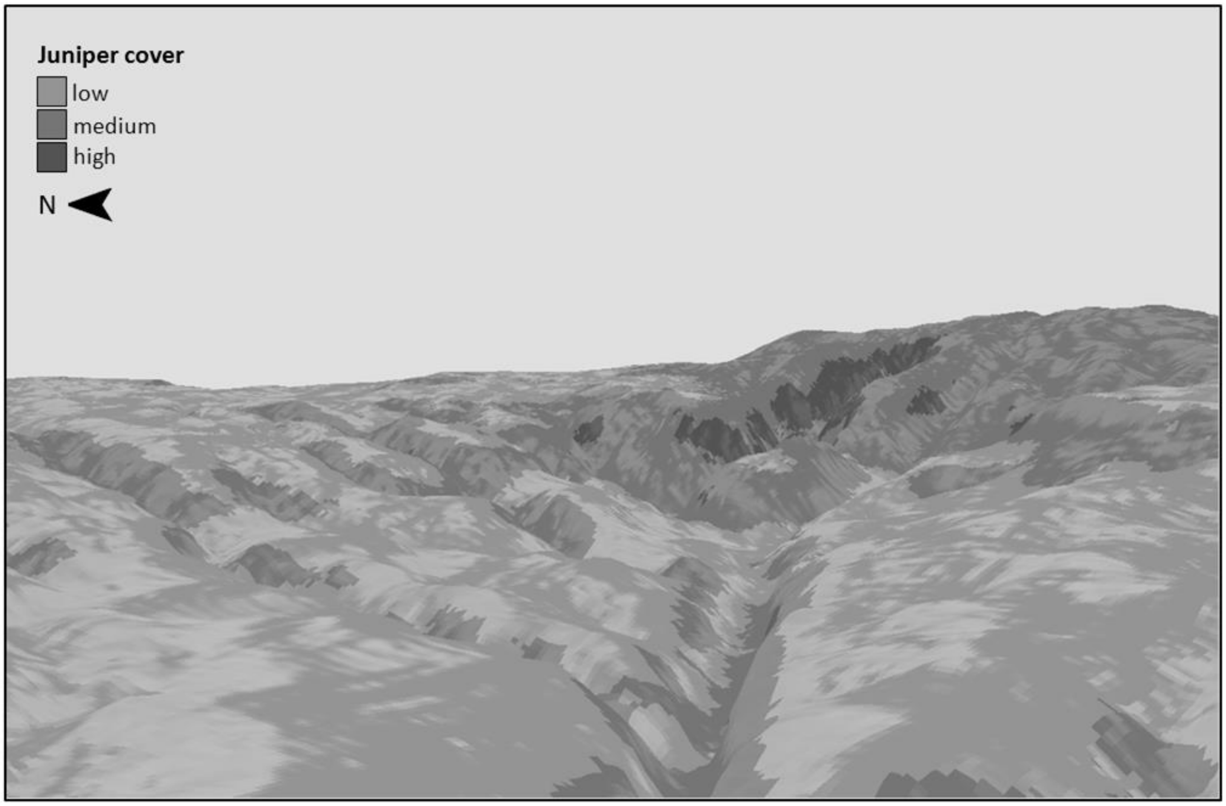

Figure 2. Model of the relationship between juniper cover at stand closure and topography characteristics applied across the study area on Juniper Mountain, Idaho. Model equation is:
Juniper cover = -49.62 + 1.69 (cos(aspect)) + 0.90 (slope) + 0.05 (elevation) – 1.32 (slope*cos(aspect)).

Management Implications

Combining the information acquired from remotely measured juniper cover and environmental/site variables has potential to be especially useful in directing juniper management priorities. Comparing estimates of juniper cover derived from NAIP imagery to potential stand closure cover values may provide a means to estimate developmental phase of juniper encroachment remotely. Identifying the developmental phase of western juniper encroachment is crucial to selecting effective management actions (Miller et al. 2005) and to prioritizing management to prevent transitions to development phases that are more costly and risky to restore. Management options become more limited and expensive as phase II woodlands transition into phase III woodlands because a reduction in understory fuel decreases the likelihood of prescribed fire carrying through the stand (Miller et al. 2000; Miller et al. 2005; Johnson and Miller 2006). Phase III stands may also be at a greater risk of exotic plant invasion following juniper control treatments than earlier phase woodlands because of a reduced herbaceous understory (Bates et al. 2014). Our results suggest that NAIP imagery and environmental/site characteristics measured from commonly available geospatial data layers have the potential to be useful in landscape scale restoration projects and land management in the Intermountain West and other ecosystems.

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Effect of Aspect on Sagebrush Steppe Recovery Post-fire in Juniper Woodlands

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SUMMARY

Restoration of sagebrush after controlling encroaching western juniper with fire in mountain big sagebrush communities is needed to improve wildlife habitat. We evaluated seeding mountain and Wyoming big sagebrush on north and south aspects after juniper control with prescribed burning. We included seeding Wyoming big sagebrush, a more drought tolerant subspecies of big sagebrush, because it might grow better than mountain big sagebrush on hot, dry south slopes or during drought. Seeding mountain big sagebrush generally increased sagebrush cover and density compared to unseeded controls and seeding Wyoming big sagebrush. Natural recovery of sagebrush was occurring on north aspects with sagebrush cover averaging 3% four years post-fire. Sagebrush was not detected on unseeded south aspects at the end of the study. Sagebrush cover and density was generally greater on north compared to south aspects, suggesting that post-fire sagebrush recovery, with and without seeding, will be variable across the landscape based on topography.

INTRODUCTION

Western juniper (*Juniperus occidentalis* ssp. *occidentalis* Hook) encroachment is one of the most prevalent issues in mountain big sagebrush (*A. tridentata* Nutt. ssp. *vaseyana* (Rydb.) Beetle) plant communities in the northern Great Basin. This ecosystem serves as an important livestock forage base and provides critical habitat for sagebrush-associated wildlife. Restoration of sagebrush communities encroached by western juniper is a priority to conserve sagebrush habitat for wildlife species (Baruch-Mordo et al. 2013) and ecosystem services (Miller et al. 2005). One of the most effective methods to control large landscapes of western juniper is prescribed burning or partial cutting (felling ¼ to ½ of mature trees to increase surface fuels) followed by prescribed burning (Bates et al. 2011; Davies et al. 2014). Burning generally results in more complete control of juniper than mechanical treatments, because mechanical treatments often fail to control juniper seedlings and small juveniles or reduce the juniper seed bank (Miller et al. 2005). Burning, however, also removes fire-intolerant sagebrush from these plant communities, which can be undesirable because sagebrush is a critical habitat component for sagebrush-associated wildlife and sagebrush-obligates, such as sage grouse, will not occupy the interior of large burns until sagebrush recovers.

Wildfires and prescribed fires generally occur prior to big sagebrush seed set; subsequently, sagebrush recruitment must occur from seed that is already at least one year old. Therefore, it may be valuable to seed mountain big sagebrush after controlling western juniper with prescribed fire. Mountain big sagebrush-dominated plant communities may also become more suited for Wyoming big sagebrush as conditions become warmer and drier with climate change. Furthermore, south aspects, generally drier and warmer than north slopes, may be less favorable to establishment of mountain big sagebrush and thus, Wyoming big sagebrush, a more drought tolerant subspecies, may establish and grow better in these environments. Wyoming big sagebrush may also establish more successfully than mountain big sagebrush at some more cool and moist locations if these locations experience a post-seeding drought.

The purpose of this research project was to investigate the effects of seeding mountain and Wyoming big sagebrush after controlling western juniper encroaching into mountain big sagebrush communities with prescribed fire on south and north aspects. We hypothesized that natural recovery of sagebrush would occur more rapidly on north than south aspects. We also expected that seeding mountain big sagebrush would expedite sagebrush recovery on north aspects more than natural recovery (unseeded) or seeding Wyoming big sagebrush, but on south aspects that seeding Wyoming big sagebrush would result in the greatest cover and density of sagebrush.

METHODS

Study sites were located in the northern Great Basin on Steens Mountain approximately 80 km southeast of Burns, OR, USA. All study sites were mountain big sagebrush-dominated plant communities prior to encroachment by western juniper. Prior to burning, the plant communities were co-dominated by western juniper and mountain big sagebrush with an understory of native perennial bunchgrasses and forbs. Juniper woodland development prior to treatment was classified as Phase II (Miller et al. 2005). Elevation at study sites was 1650-1775 m above sea level. Aspects of study sites were north and south. Slopes ranged between 30 and 35%. South and noth aspects were South Slopes 12-16 PZ (R023XY302OR) and North Slopes 12-16 PZ (R023XY31OR) Ecological Sites, respectively. Both aspects had a frigid temperature regime and xeric moisture regime. Long-term average annual precipitation (1981-2010) was 405 mm with the majority occurring during the cool season. Annual precipitation was 89, 87, 63, and 87% of the long-term average in 2011, 2012, 2013, and 2014. Livestock were excluded for one year prior and one year post-burning on all study sites.

The effects of seeding different subspecies of big sagebrush on north and south aspects that had been prescribed burned to remove encroaching juniper were evaluated using a split-plot design with four complete replicates of all treatments on each aspect. Treatments included an unseeded control, seeded with mountain big sagebrush, and seeded with Wyoming big sagebrush, and were randomly assigned to three plots in each block on each aspect. All treatment plots were prescribed burned in late September of 2011 using head-fires ignited with drip torches. All fires were complete burns resulting in 100% morality of juniper and sagebrush plants. Sagebrush seed was broadcast seeded with a handheld seeder at 500 PLS$\cdot$m^{-2} in November of 2011. Plant community characteristics were measured in July of 2013, 2014, and 2015.

RESULTS

Density and cover of herbaceous vegetation generally did not differ among treatments, but did significantly differ between aspects. Perennial herbaceous vegetation cover and density was generally greater on north compared to south aspects. Exotic annual grass cover and density was greater on south compared to north aspects. Sagebrush density was, on average, more than 40 times greater on the north compared to south aspects. Sagebrush density was greater in all years on both aspects in mountain big sagebrush seeded plots compared to unseeded control plots (Fig. 2). Sagebrush density on Wyoming big sagebrush seeded plots generally did not differ from unseeded control plots in most years, except it was greater than the controls on the north aspect in 2015 (Fig. 2). Sagebrush density was similar in mountain and Wyoming big sagebrush seeded plots in most years on both aspects, except for on the north aspect in 2015 when sagebrush density was greater in mountain big sagebrush seeded plots compared to Wyoming big sagebrush seeded plots (Fig. 2). Natural recovery of sagebrush density was not occurring on south aspects (Fig. 2), while some sagebrush was detected on north aspects in unseeded controls (Fig. 2). Sagebrush cover on north aspects in Wyoming big sagebrush seeded plots did not differ from controls in any year and mountain big sagebrush seeded plots in 2013 and 2014 (Fig. 3). In 2015, sagebrush cover on north aspects were greater in mountain big sagebrush compared to Wyoming big sagebrush seeded plots (Fig. 3). Sagebrush cover increased with time on north aspects, but did not vary with time on south aspects. Sagebrush cover was similar among treatments in all years on south slopes, except it was greater in mountain big sagebrush seeded plots compared to control plots by 2015. Total shrub density and cover followed a pattern similar to sagebrush density and cover.

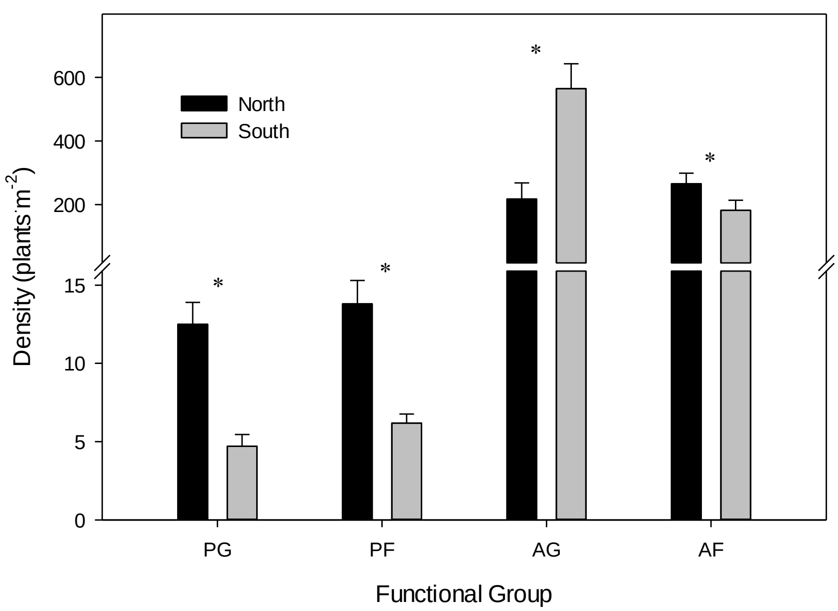

Figure 1. Herbaceous functional group density (mean + SE) by aspect summarized across treatments and years (2013-2015). PG = perennial grasses, PF = perennial forbs, AG = annual grasses, and AF = annual forbs. Asterisks (*) indicates significant difference ($P \leq 0.05$) between aspects for that functional group.

DISCUSSION

Seeding mountain big sagebrush after controlling western juniper with prescribed fire accelerated the recovery of sagebrush cover and density. Similar results were reported by Davies et al. (2014) when they seeded mountain big sagebrush in combination with perennial grasses and forbs after juniper control with partial cutting followed by prescribed burning. Davies et al.

(2014) also reported wide-ranging levels of success with sagebrush cover varying from 1% to 12% among seeded sites by the third year post-seeding. In agreement, we found recovery of sagebrush cover and density varied considerably among sites seeded with mountain big sagebrush. The majority of the variability in sagebrush cover and density in our study was related to aspect. For example, sagebrush cover in mountain big sagebrush seeded plots was 19 times greater on north compared to south aspects in the final year of study. Though sagebrush cover and density on south aspects seeded with mountain big sagebrush were low, cover and density were greater than the unseeded control at the conclusion of the study. This suggests that rapid recovery of sagebrush after fire on these juniper-encroached south aspects is unlikely, though seeding mountain big sagebrush does hasten sagebrush recovery.

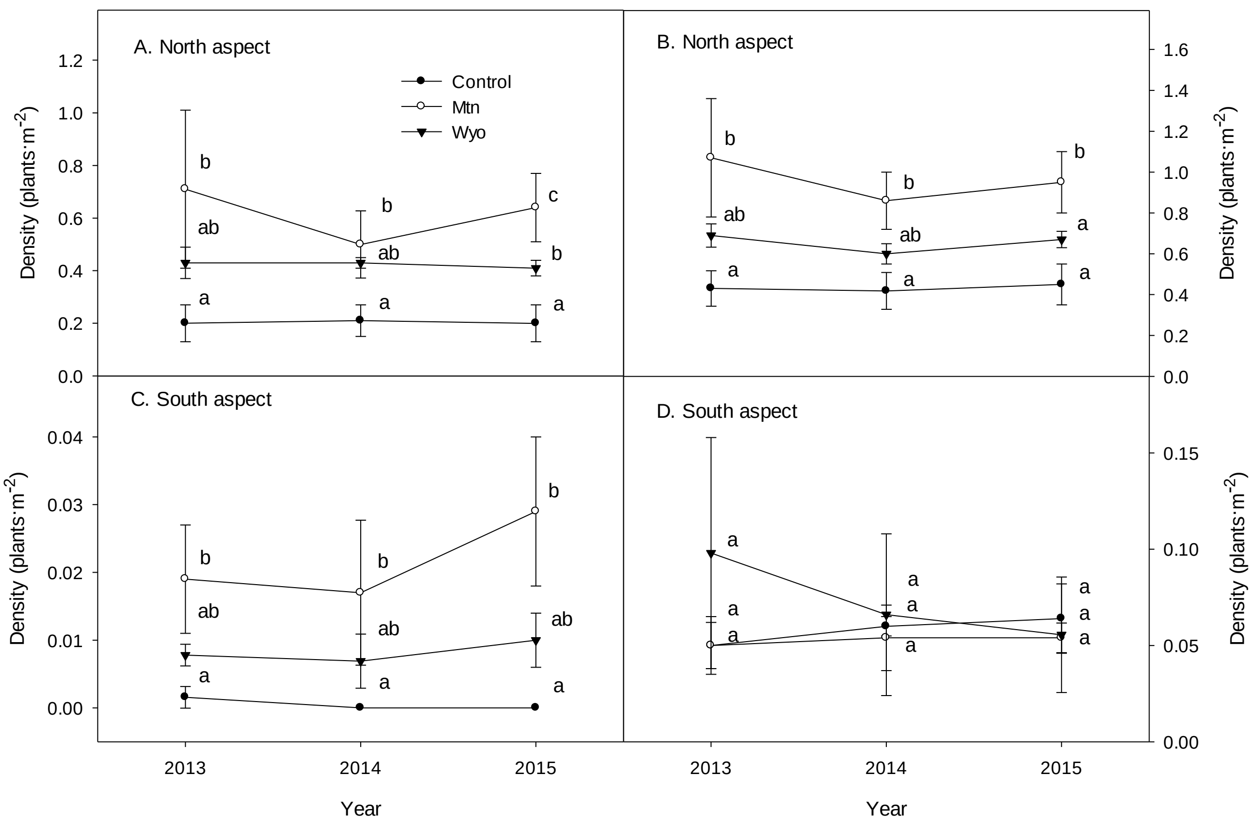

Figure 2. Sagebrush (A & C) and total shrub (B & D) density (mean ± SE) in treatments on north aspects and south aspects in 2013, 2014, and 2015. Control = unseeded control, Mtn = mountain big sagebrush seeded, and Wyo = Wyoming big sagebrush seeded. Different lower case letters signify differences ($P \leq 0.05$) between treatments in that year. Scale varies by figure panel.

Natural recovery of sagebrush was occurring on north slopes with approximately 3% sagebrush cover by the conclusion of the study. Dissimilar to north aspects, there was no evidence of natural recovery of big sagebrush on south aspects. At the end of the study, sagebrush was not detected in unseeded plots on south aspects; suggesting that natural sagebrush recovery on south slopes will be slow. Our research suggests that these two vastly different recovery trajectories may occur in the same burned landscape based on the influence of landscape characteristics on the seedling establishment environment. Our results did not provide any evidence that seeding Wyoming big sagebrush would be advantageous on sites formerly occupied by mountain big sagebrush.

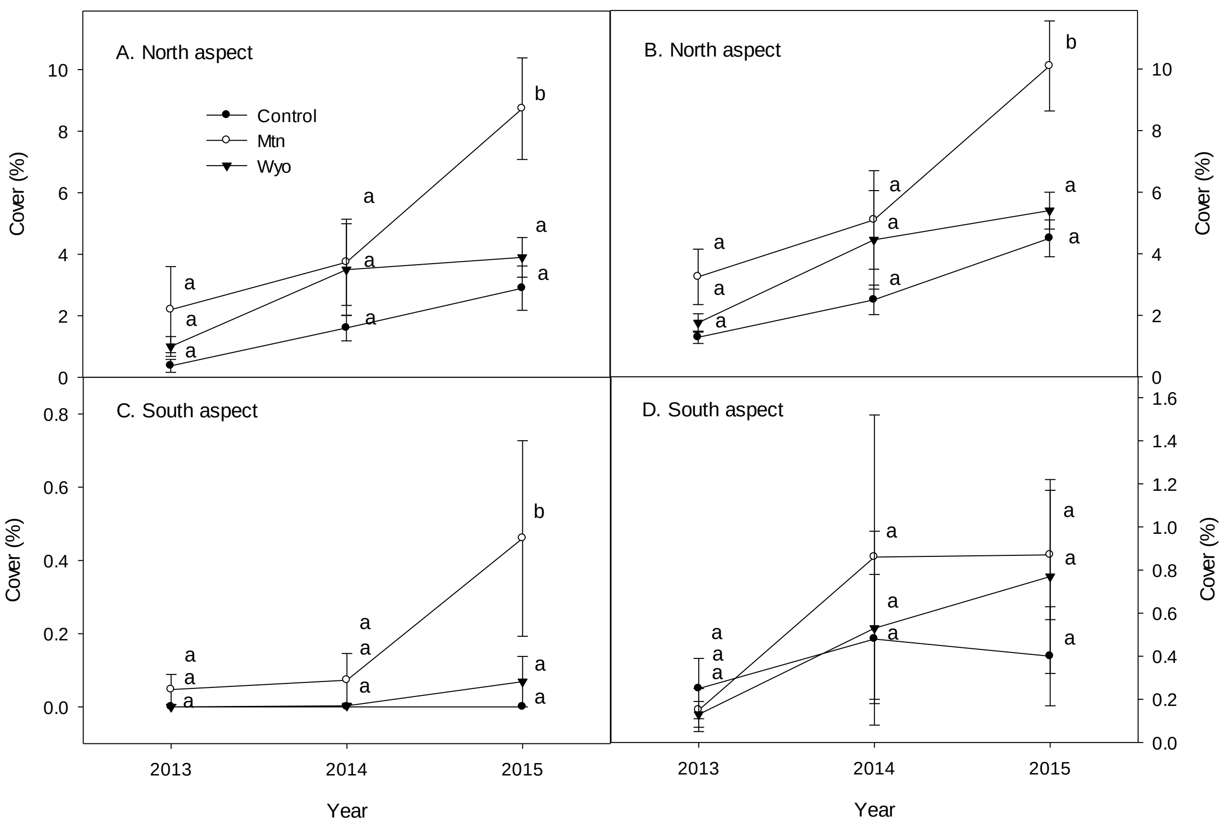

Figure 3. Sagebrush (A & C) and total shrub (B & D) cover (mean ± SE) in treatments on north aspects and south aspects in 2013, 2014, and 2015. Control = unseeded control, Mtn = mountain big sagebrush seeded, and Wyo = Wyoming big sagebrush seeded. Different lower case letters signify differences ($P \leq 0.05$) between treatments in that year. Scale varies by figure panel.

MANAGEMENT IMPLICATIONS

Topography dictates vegetation recovery both with and without seeding and this suggests that landscapes should be divided into topographically similar units for restoration efforts. Clearly sagebrush steppe restoration after fire will be more successful on north compared to south aspects. However, seeding mountain big sagebrush after fire controlled western juniper accelerated sagebrush recovery on both south and north aspects. These results suggest that sagebrush habitat can be restored by prescribed burning encroaching juniper followed by broadcast seeding mountain big sagebrush.

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Restoring Mountain Big Sagebrush Communities after Prescribed Fire in Juniper Encroached Rangelands

Kirk W. Davies, Jon D. Bates, Matt Madsen, Aleta Nafus

SUMMARY

Western juniper encroachment into sagebrush steppe communities has reduced livestock forage production, increased erosion and runoff risk, and degraded sagebrush-associated wildlife habitat. We evaluated seeding perennial herbaceous vegetation and sagebrush at five sites where juniper was controlled with prescribed fire. Results suggest that broadcast seeding perennial herbaceous vegetation can accelerate perennial grass recovery and stabilize the site. Our results also demonstrated that seeding mountain big sagebrush after prescribed burning juniper can rapidly recover sagebrush cover and density. Where sagebrush habitat is limited, broadcast seeding sagebrush after juniper control can rapidly recover sagebrush habitat for sagebrush-associated species.

INTRODUCTION

Mountain big sagebrush (*Artemisia tridentata* spp. *vaseyana* (Rydb.) Beetle) plant communities are being encroached by juniper (*Juniperus* L.) and piñon pine species (*Pinus* L.). In the northern Great Basin and Columbia Plateau, western juniper (*Juniperus occidentalis* ssp. *occidentalis* Hook) has increased from 0.3 million ha to 3.5 million ha since the 1870's (Miller et al. 2000). As juniper cover increases, sagebrush and other shrubs are lost, herbaceous diversity and biomass production decreases, and runoff and erosion potential increases (Miller et al. 2005). Western juniper encroachment is detrimental to sagebrush obligate wildlife species because of the loss of sagebrush, decreases in herbaceous vegetation, and increased predation risk.

The most cost-effective method to control encroaching western juniper is prescribed burning, but sagebrush and perennial grass recovery may be slow. Sagebrush recovery may be exceptionally slow because sagebrush is excluded from the plant community with juniper dominance. The purpose of this research was to determine if the recovery of mountain big sagebrush plant communities after juniper control with fire could be expedited by broadcast seeding perennial herbaceous vegetation and mountain big sagebrush. We hypothesized that natural recovery of sagebrush in juniper dominated plant communities is constrained by limited sagebrush seed in the seed bank and thus, seeding sagebrush would significantly accelerate sagebrush recovery.

METHODS

The study was conducted on Steens Mountain in southeastern Oregon approximately 80 km southeast of Burns, OR (lat 42˚ 33' 36"N, long 118˚ 19' 12" W). Prior to prescribed burning, the plant communities were dominated by western juniper (late Phase II and Phase III) with an understory of perennial grasses and forbs. Elevation among study sites ranged from 5728 to 5932 ft above sea level.

At five different sites we applied the following treatments: unseeded control (CONTROL), seeded with perennial herbaceous vegetation (SEED), and seeded with perennial herbaceous vegetation and mountain big sagebrush (SEED+SAGE). Perennial herbaceous vegetation was aerially broadcast seeded the first week of November 2009 using a fixed wing aircraft. Mountain big sagebrush was broadcast seeded with a hand-cranked broadcaster to simulate aerial seeding immediately after herbaceous seeding. The perennial herbaceous seed mix consisted of Idaho fescue, Sherman big bluegrass, Oahe intermediate wheatgrass, Manchar smooth brome, Paiute orchardgrass, Maple Grove Lewis flax, and Ladak alfalfa. Vegetation cover and density was measured in July of the first, second, and third years (2010, 2011, and 2012) after seeding. Sagebrush cover was also measured in 2013 and 2014 to provide a longer term evaluation of sagebrush recovery.

RESULTS

By the third year post-seeding, large perennial grass cover was 2.0 and 2.5-fold greater in the SEED+SAGE and SEED treatments compared to the CONTROL treatment, respectively (Fig. 1A; $P = 0.03$ and < 0.01, respectively). Similarly, perennial grass density was 1.7- and 2.2-fold greater in the SEED+SAGE and SEED treatments compared to the CONTROL treatment (Fig. 2A; $P = 0.02$ and < 0.01, respectively). The SEED+SAGE treatment had greater sagebrush cover than the SEED and CONTROL treatments ($P = 0.03$ and 0.02, respectively) and continued to increase (Fig. 3). Sagebrush cover increased in the SEED+SAGE treatment over time, but remained relatively unchanged in the SEED and CONTROL treatments (Fig. 1B; $P < 0.01$). In 2012, sagebrush cover was 74- and 290-fold greater in the SEED+SAGE treatment compared to the CONTROL and SEED treatments, respectively. In 2012, sagebrush density was 62- and 155-fold greater in the SEED+SAGE treatment compared to the SEED and CONTROL treatments. Sagebrush density increased in the SEED+SAGE treatment almost 10-fold between the first and second year after seeding, but remained unchanged in the SEED and CONTROL treatments.

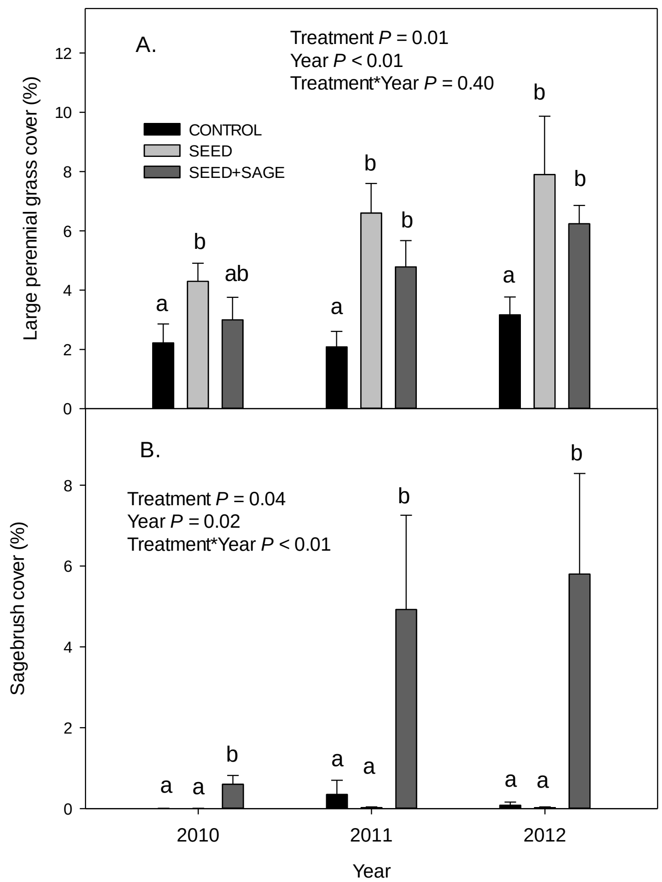

Figure 1. Large perennial grass (A) and sagebrush (B) cover after partial cutting and prescribed burning western juniper encroached mountain big sagebrush communities that were not seeded (CONTROL), seeded with a herbaceous seed mix (SEED), or seeded with a herbaceous seed mix plus sagebrush (SEED+SAGE). Different letters indicate a significant difference ($P \leq 0.05$) between treatments in that year.

DISCUSSION

Aerial seeding after juniper control accelerated herbaceous vegetation recovery by approximately doubling large perennial grass cover and density. Rapid recovery of this functional group likely stabilizes the plant community as established perennial grasses can greatly limit exotic plant invasion and decrease erosion and runoff risk after juniper control. Our results suggest that natural recovery of the herbaceous understory will probably occur, but may be slow.

The lack of a perennial forb response with seeding was probably the result of limited establishment of seeded species. No alfalfa was detected in any of the treated plots and only a few Lewis flax plants were found in each plot. There were some places outside of the sampling plots that were aerially seeded where alfalfa and Lewis flax established at higher densities.

The lack of a treatment effect on exotic annual grass cover and density was probably due to most sites not having a significant annual grass presence regardless of treatment. Therefore, our study was not a robust test for determining the efficacy of aerial seeding for limiting exotic annual grasses. When only evaluating the two sites that had a significant annual grass presence (>0.5% cover), annual grass cover and density were 2.7- and 3.8-fold greater in the unseeded (9.3±1.0% and 281±63 plants·m^{-2}) compared to the aerially seeded (3.5±1.0% and 73±21 plants·m^{-2}) areas in the third year after treatment, respectively. These results suggest that it may be important to seed after juniper control where exotic annual grasses are a threat.

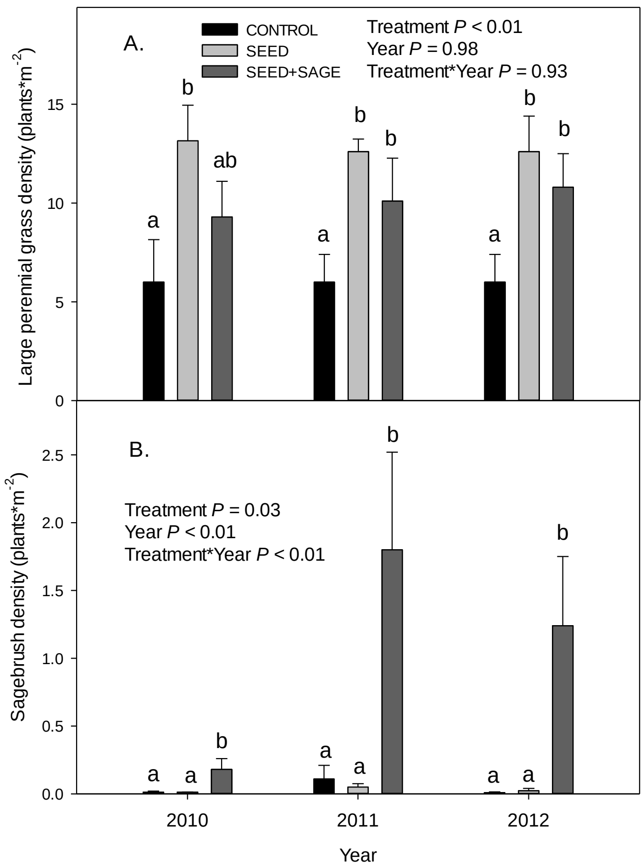

Figure 2. Large perennial grass (A) and sagebrush (B) density after partial cutting and prescribed burning western juniper encroached mountain big sagebrush communities that were not seeded (CONTROL), seeded with a herbaceous seed mix (SEED), or seeded with a herbaceous seed mix plus sagebrush (SEED+SAGE). Different letters indicate a significant difference ($P \leq 0.05$) between treatments in that year.

Our results suggest that seeding mountain big sagebrush after using prescribed fire to control western juniper can greatly accelerate the recovery of sagebrush cover and density. By the fifth year after treatment, sagebrush cover averaged 18% in the sagebrush seeded plots. Late Phase II and Phase III juniper woodlands have largely excluded sagebrush from the plant communities; therefore, seed input would be limited. Thus, these plant communities are likely sagebrush seed limited without seeding.

We observed seeded sagebrush plants producing seed by the second year after seeding. This suggests that the sagebrush seeded areas could serve as a seed source for unseeded areas as well as providing additional recruitment potential in seeded areas. We speculate that even lower sagebrush seeding rates than used in this study may be successful because seeded sagebrush will start producing seed in a few years. However, lower rates may increase the length of time for sagebrush recovery. In contrast, herbaceous recovery may be accelerated with lower sagebrush abundance.

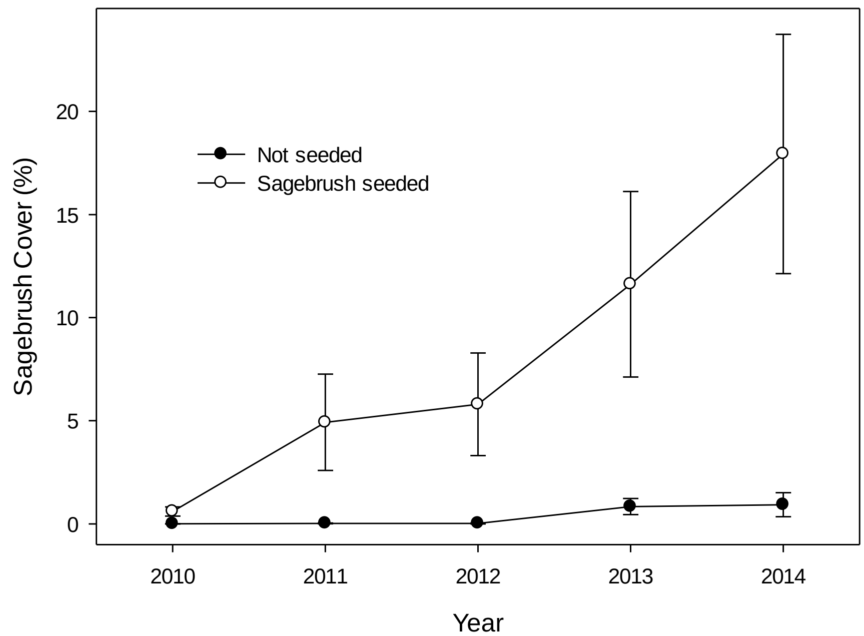

Figure 3. Sagebrush cover after prescribed burning western juniper encroached mountain big sagebrush communities that were not seeded or seeded with sagebrush.

MANAGEMENT IMPLICATIONS

Our results suggest that seeding mountain big sagebrush after prescribed burning western juniper can improve sagebrush-associated wildlife species habitat. Our research suggests that western juniper encroached sagebrush steppe similar to our study area under similar climatic conditions may be restored in a relatively short time period with western juniper control followed by seeding sagebrush (Davies et al. 2014). These results also suggest that seeding herbaceous species and sagebrush after prescribed burning can limit opportunities for invasive plants. Seeding herbaceous vegetation, however, may not always be needed. Our results suggest that sagebrush recovery with seeding may be adequate to provide sage-grouse habitat at some sites in a few years after prescribed burning western juniper.

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Sagebrush Steppe Recovery after Fire Varies by Successional Phase of Western Juniper Woodland

Jon D. Bates, Kirk W. Davies

SUMMARY

Western juniper (*Juniperus occidentalis* Hook.) in North America are encroaching other plant communities because of a reduced in fire frequency. Prescribed fire is being increasingly employed to restore juniper-encroach sagebrush steppe communities. We compared vegetation recovery following prescribed fire on Phase 2 (mid-succession) and Phase 3 (late-succession) western juniper woodlands. The Phase 2 site maintained native herbaceous species before and after fire. The Phase 3 site shifted from native herbaceous species to dominance by invasive weeds. The results suggest that Phase 2 sites are more likely to recover native vegetation after fire, while indicating sites transitioning from Phase 2 to Phase 3 cross a recovery threshold where the potential is greater for invasive weeds to dominate.

INTRODUCTION

Western juniper are encroaching sagebrush steppe plant communities because of a reduced in fire frequency. Adverse effects of woodland expansion on sagebrush steppe communities include loss of wildlife habitat, elimination of shrubs, and reduced herbaceous diversity/productivity. Thus, woodland control in sagebrush steppe, mainly using fire, has been a major management focus. However, forecasting vegetation recovery following prescribed fire is less predictable as western juniper woodland development varies across landscapes. Within Phase 1 woodlands, shrubs and herbaceous species are the dominant vegetation with few trees present; in Phase 2 woodlands, trees co-dominate with shrubs and herbaceous plants; and in Phase 3 woodlands, trees are dominant and shrubs and herbaceous layers are reduced. The transition from Phase 2 to Phase 3 woodlands alters fuel characteristics and this likely changes fire behavior and increases fire severity, leading to a post-fire risk of weed dominance. Thus, we expect that Phase 3 woodlands may have crossed a threshold, where natural recovery is uncertain and additional inputs may be required to restore sagebrush steppe communities. However, this has not been tested. Our objective was to identify transition thresholds for recovering sagebrush steppe vegetation by comparing the recovery of the mountain big sagebrush (*Artemisia tridentata* Nutt. ssp. *vaseyana* (Rydb.) Beetle) steppe herbaceous community after prescribed fire in Phase 2 and Phase 3 western juniper woodlands.

METHODS

The study was located in Kiger Canyon, Steens Mountain, southeastern Oregon. Twelve Phase 2 plots and nine Phase 3 plots, all measuring about 1.5 acres, were established in May 2003. Criteria for determining woodland phase (cover of herbaceous, shrub and tree life forms) were taken from Miller et al. (2005). Phase 2 and Phase 3 woodlands were intermixed within an area of 15 km^2 and were independent of each other (Fig. 1).

Cutting involved felling 1/3 of the dominant and sub-canopy western juniper trees (>0.3m tall) in an even distribution throughout the stands. Trees were cut and dried over the summer, followed by fall burning in 2003. Recovery depended on natural succession and no post-fire seeding was undertaken. Livestock were excluded for 2 years before burning to increase fine fuel loads. Vegetation characteristics were measured in June (2003–2007, 2009) and July (2012). Livestock grazed intermittently in the post-fire years with low to moderate utilization.

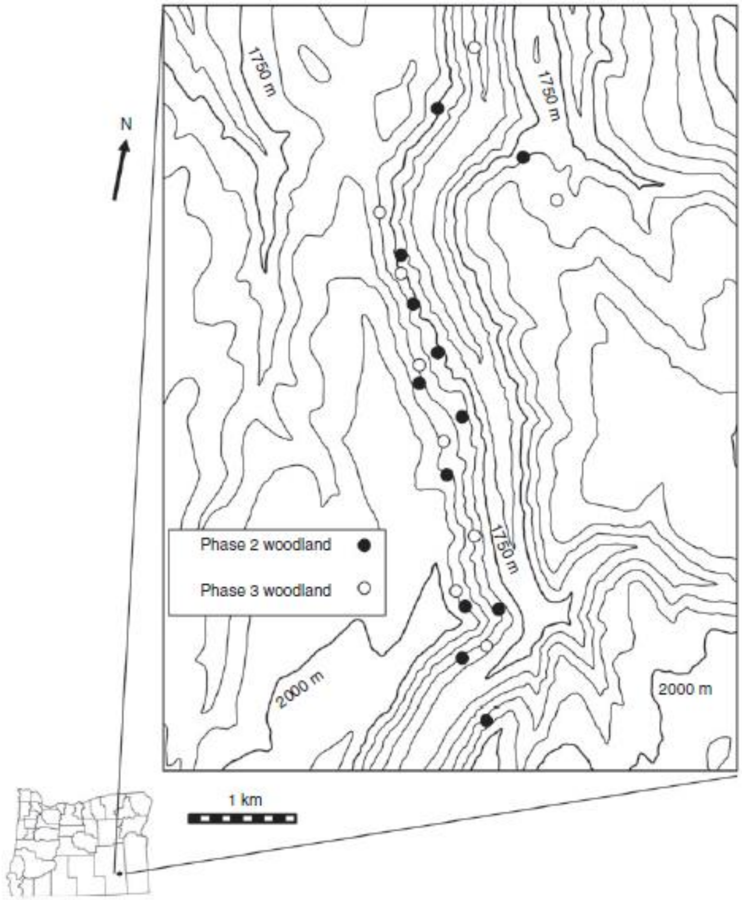

Figure 1. Phase 2 and Phase 3 woodland sites for the Kiger Canyon study area, Steens Mountain, Harney County, Oregon, USA. Phase 2 woodlands represent a co-dominance of trees, shrubs and herbaceous plants and in Phase 3 woodlands, trees are dominant and shrubs and herbaceous layers are reduced.

RESULTS

The prescribed fires killed remaining uncut western juniper trees in both Phase 2 and 3 woodland sites. The fire also consumed all fuels up to and including 1000-h fuels.

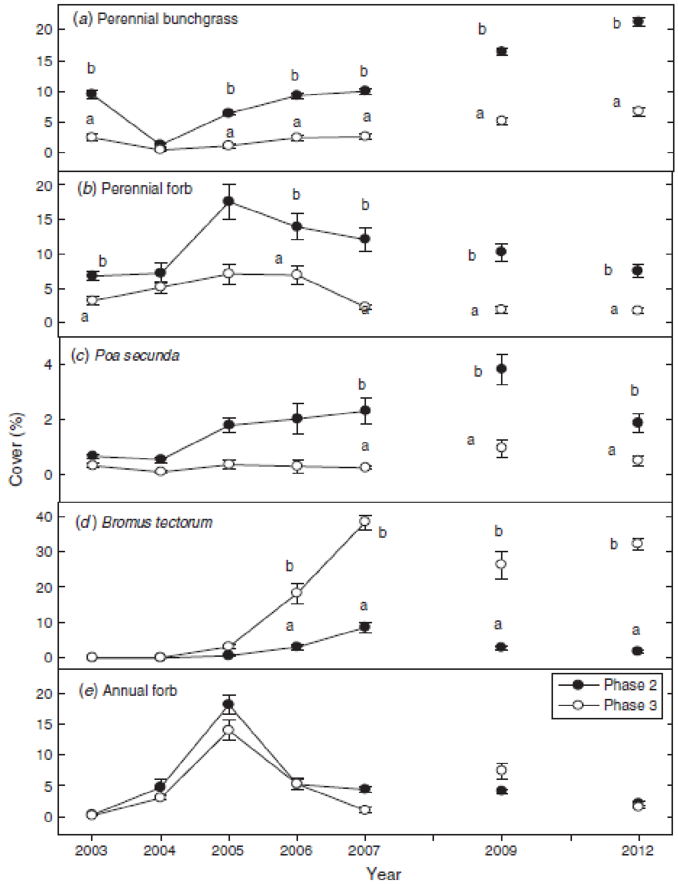

Figure 2. Functional group cover (%) in burned Phase 2 and Phase 3 western juniper woodlands, Steens Mountain, Oregon (2003–2012; 2003 is the pre-fire year): (a) perennial grasses; (b) perennial forbs; (c) Poa secunda; (d) Bromus tectorum and (e) annual forbs. Data are means ±1 standard error. Means sharing a common lowercase letter are not significantly different ($P > 0.05$).

Prior to fire, perennial grass cover was 4 times ($P < 0.0001$) greater in Phase 2 sites than in Phase 3 sites (Fig. 2a). From 2004 to 2012 perennial grass cover was 3–6 times greater ($P < 0.0001$) in Phase 2 than in Phase 3 sites. Perennial forb cover was 2 times ($P < 0.0001$) greater in Phase 2 than in Phase 3 sites before treatment (Fig. 2b). After fire, perennial forb cover was 2–10 times greater ($P < 0.001$) in Phase 2 than in Phase 3 sites. Cheatgrass was present in trace amounts before treatment in both woodland phases, but increased significantly after fire (Fig. 2d; $P < 0.0001$). Cheatgrass cover was 4–16 times greater in the Phase 3 than the Phase 2 sites, in 2006–2012 ($P < 0.0001$).

Before fire, perennial grass density was 3 times greater in Phase 2 than in Phase 3 sites (Fig. 3a; $P < 0.0001$). Burning decreased perennial grass density by 78% in the Phase 2 sites, from ~14 to 2–3 plants·m^{-2}. Phase 3 sites showed a decline of 95% in perennial grass density, from ~4 to <1 plants·m^{-2} ($P = 0.004$). Perennial grass densities have increased in both phases since fire, but from 2005 to 2012 densities were 4–5 times greater in the Phase 2 sites ($P < 0.0001$). Densities of perennial forbs were 4–5 times greater in the Phase 2 than Phase 3 sites after fire (Fig. 3b; $P = 0.002$).

DISCUSSION

Prescribed fire in two different phases of woodland development provided a distinct contrast in herbaceous recovery in western juniper invaded sagebrush steppe. The first two years after fire herbaceous recovery was mainly comprised of perennial and annual forbs on both burned woodland phases, a successional stage typical of juniper woodlands following fire (Bates et al. 2011). However, by the third year after fire, vegetation succession had diverged between phases, with cheatgrass dominating Phase 3 sites and herbaceous perennials dominating Phase 2 sites. Cheatgrass was dominant on Phase 3 sites even after perennial grasses had returned to pre-burn levels of cover and density by the fourth and sixth year after fire, respectively.

Increasingly, experimental evidence indicates that the resilience of mountain big sagebrush steppe communities following fire is dependent on the persistence of sufficient density of perennial herbaceous vegetation (Bates et al. 2011; Condon et al. 2011). The limited increase in cheatgrass on Phase 2 sites were likely due to the persisting density of perennial grasses and forbs first year post-fire, and near full recovery by the fourth year post-fire. The greater presence and recovery of perennial herbaceous vegetation has been indicated by others to prevent annual grasses from dominating after fire in sagebrush steppe (Davies et al. 2008; Condon et al. 2011).

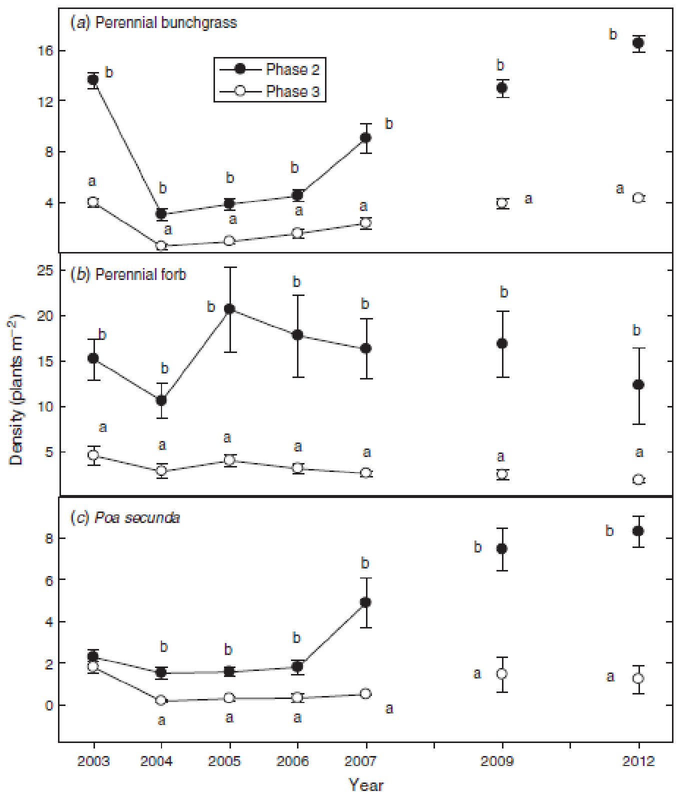

Figure 3. Herbaceous perennial densities (plantsm _2) in burned Phase 2 and Phase 3 western juniper woodlands, Steens Mountain, Oregon (2003–2012; 2003 is the pre-fire year): (a) perennial grasses; (b) perennial forbs; (c) Poa secunda. Data are means ± 1 standard error. Means sharing a common lowercase letter are not significantly different (P > 0.05).

Despite dominance by cheatgrass in post-fire Phase 3 woodland, perennial grass density and cover continued to increase. Should this trend continue, native species may, over a longer period, replace cheatgrass. However, the current dominance by cheatgrass leaves potential for this species to alter the fire regime and limit native species recovery.

Our results suggest that sites in Phase 1 and Phase 2 woodland encroachment will likely recover following fire disturbances. Phase 3 woodlands may recover following fire or be invaded by exotics, with surviving perennial plant density and invasive species presence becoming the prime determinants of plant community succession. We suggest that native species composition will recover in Phase 2 and

Phase 3 woodlands when perennial grass and forb densities respectively exceed 1 and 5 plants/yd^2 post-fire, based on results from Bates et al. (2011). Sites with herbaceous values below these levels, as in our study, have a greater risk of becoming dominated by invasive annual grasses following fire, indicating that a threshold may have been crossed.

CONCLUSIONS

Burning in Phase 3 woodlands is less predictable because of depleted understories and severe fire effects on herbaceous vegetation, which increase the risk of post-fire weed dominance. Phase 3 woodlands that are burned by wild or prescribed fire are more likely to require additional inputs, primarily seeding and weed control, for vegetation recovery goals to be realized. Priority should be given to treat Phase 1 and Phase 2 woodlands before they transition to Phase 3. Phase 1 and 2 woodlands, which have an intact understory of shrubs and herbaceous species, will most likely be dominated by native vegetation after fire. It will take several decades for mountain big sagebrush to recover following burning of Phase 1 and 2 woodlands; however, there is greater potential for achieving recovery goals and preventing woodland dominance by reintroducing fire in Phase 2 and earlier stages of juniper woodland development.

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Herbaceous production response to juniper treatment

Jon Bates, Kirk W. Davies, and Tony Svejcar

SUMMARY

Western juniper (*Juniperus occidentalis* Hook.) has expanded and infilled into other plant communities the past 130 to 150 years in the semi-arid Pacific Northwest. The increase in juniper reduces herbaceous forage and browse provided by shrubs for livestock and big-game. We measured herbaceous production in a variety of plant communities following cutting or prescribed fire treatments in Phase 1 and 2 (early to mid-succession) and Phase 3 (late-succession) western juniper woodlands. Herbaceous production increased but results varied depending on site potential and woodland removal method. Woodland cutting resulted in herbaceous production increases of 50% (Phase 2) up to 700% (Phase 3). Prescribed fire resulted in 2 to 3-fold increases in herbaceous production in Phase 1 and 2 woodlands and 3 to 10-fold in Phase 3 woodlands.

INTRODUCTION

Western juniper encroachment into sagebrush steppe plant communities has a number of adverse effects including greater soil erosion, loss of wildlife habitat, and reduced herbaceous and shrub productivity. Control of western juniper woodlands has mainly been by prescribed fire and cutting. Following juniper control, measurement of herbaceous production has generally been limited to short-term studies (less than 3-5 years) with no comparison across sites. Here we gathered herbaceous production data, from multiple sites, and for time periods between 4 to 24 years after woodland treatment.

METHODS

Studies were located on Steens Mountain, Hart Mountain, and the Northern Great Basin Experimental Range (NGBER), SE Oregon and South Mountain, SW Idaho. Studies took place between 1992 and 2015 and were located in mountain, Wyoming, and basin big sagebrush steppe plant communities (Table 1). Treatments were cutting with trees left on site, cutting with winter burning of trees in place, and prescribed (Rx) fire. For detailed juniper treatments contact authors or refer to associated references.

At the study sites, standing crop biomass was determined by herbaceous lifeform group by clipping 15 to 20, 1 m^2 frames per treatment plot in late May to mid-June. Lifeforms were *P. secunda* (Sandberg's bluegrass), perennial bunchgrasses (e.g., Idaho fescue, Thurber's needlegrass, and bluebunch wheatgrass), cheatgrass, perennial forbs, and annual forbs. Perennial bunchgrasses were clipped to 2-cm stubble height. Other functional groups were clipped to near ground level. Harvested herbage was dried at 48^o C for 72 hours prior to weighing. Production of perennial grasses and *P. secunda* was determined by separating current year's growth from standing crop. Standing crop biomass includes current year's growth and standing residual biomass remaining from previous year~~’~~s’ growth. Standing crop of perennial forbs, annual forbs, and cheatgrass were equivalent with their annual yields and required no separations. Repeated measures using the PROC MIXED procedure (SAS 9.3) for a randomized complete block design was used to test year, treatment, and year by treatment effects for herbaceous response variables. Statistical significance of all tests were set at $P<0.05$. Data provided show production values after site recovery and had largely stabilized unless otherwise noted.

RESULTS

Herbaceous production increased in response to cutting and prescribed fire of woodlands, though the level of response was influenced by site potential and type of woodland treatment. Perennial grasses and forbs tended to be the dominant increasers, though in several cases cheatgrass was either a major contributor or dominated following juniper control. Annual forbs and Sandberg’s bluegrass tended to increase in the first 2 to 4 years after tree control after which treatment differences mostly disappeared.

At site 1 (Hart Mt.) prescribed burning nearly tripled perennial bunchgrass production and doubled total herbaceous production compared to unburned controls (Table 1). Perennial grass production peaked sometime between the 4 and 7th year after fire. Total herbaceous production increased immediately and in the first 4 years after fire forbs represented a greater percentage of the total. At the NGBER (site 2A and 2B) it required 3 years before herbaceous production (perennial lifeforms and total) in prescribed fire and cut/winter burn treatments increased above controls (Table 1; Fig. 1). Prescribed fire was more effective than cut/winter burn treatments at increasing perennial grass (Rx fire 2.5 to 3-fold increased; cut/winter burn 2-fold increase) and total herbaceous production (Rx fire 2-fold increase; cut/winter burn 1.5-fold increase) compared to controls. On Steens Mountain (site 3), plant community and treatment method determined post treatment response and herbaceous composition. At the mountain big sagebrush/Idaho fescue community (site 3A) prescribed fire was effective at increasing perennial grass (3.5-fold), perennial forbs (2-fold) and total herbaceous production (2.7-fold) above controls. These differences remained consistent between 2009 and 2014. At the basin big sagebrush/bluebunch wheatgrass associations (3B-D) treatments were effective at increasing herbaceous production but because these sites have a mesic soil temperature regime cheatgrass was a major contributor to site productivity (Table 1). Prescribed fire at site 3B resulted in a 10-fold increase in perennial bunchgrass, a 30-fold increase in cheatgrass and a 4-fold increase in total herbaceous production compared to the controls. Perennial bunchgrasses comprised about 50% of total production and cheatgrass 30% of total production.

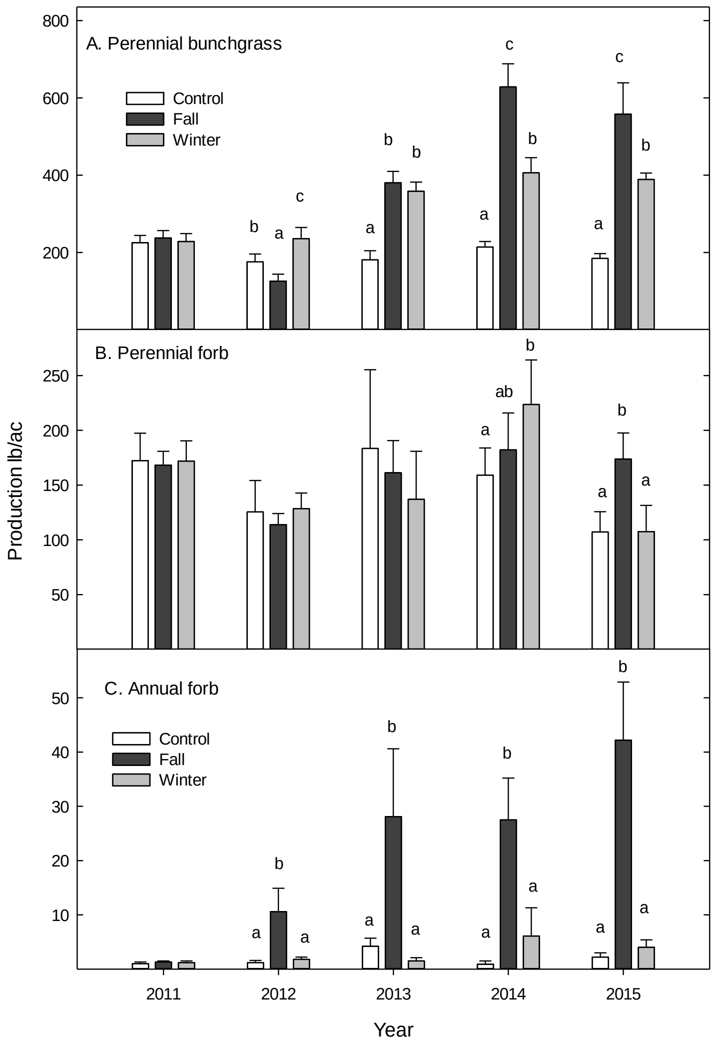

Figure 1. Herbaceous production (lb/ac), mountain big sagebrush/ Idaho fescue site, NGBER (2A site), 2011-2015, for A) perennial bunchgrasses, B) perennial forbs, and C) annual forbs in cut and control woodlands. Fall burn was done Sep 2011. Winter was cut July 2011 and burned Jan 2012. Different lower case letters indicate differences between treatments within each year for each life form. The pretreatment year is 2011.

Prescribed fire at site 3C replicated this trend, however, because the annual forb component was largely invasive species, native perennial lifeforms comprised about 60% and invasive annuals about 40% of total herb production. Perennial bunchgrasses at site 3C were about 2-fold and 9-fold greater in the fire treatment, respectively, than the cut/winter burn treatment and the control. Interestingly annual grasses were 2 fold greater in the cut/winter burn than the prescribed fire treatment. Annuals in the cut/winter burn treatment comprised about 70% of total herbage.

Site 3D has records spanning 24 years after juniper cutting and herbaceous composition is dynamic over time (Table 1; Fig. 2). Through most of these years herbaceous perennial lifeforms and total production were 5 to 10-fold greater in the cut compared to the control. Herbaceous production peaked and remained fairly stable between 1997 and 2009. Since 2003-2004, native species production comprised about 80% of total herbage. What has become obvious is that since 2009 herbage production has declined by half since 2005-2007 as juniper and shrub cover has increased. Juniper (3.1%) and shrub (5%) cover totaled over 8% in 2015.

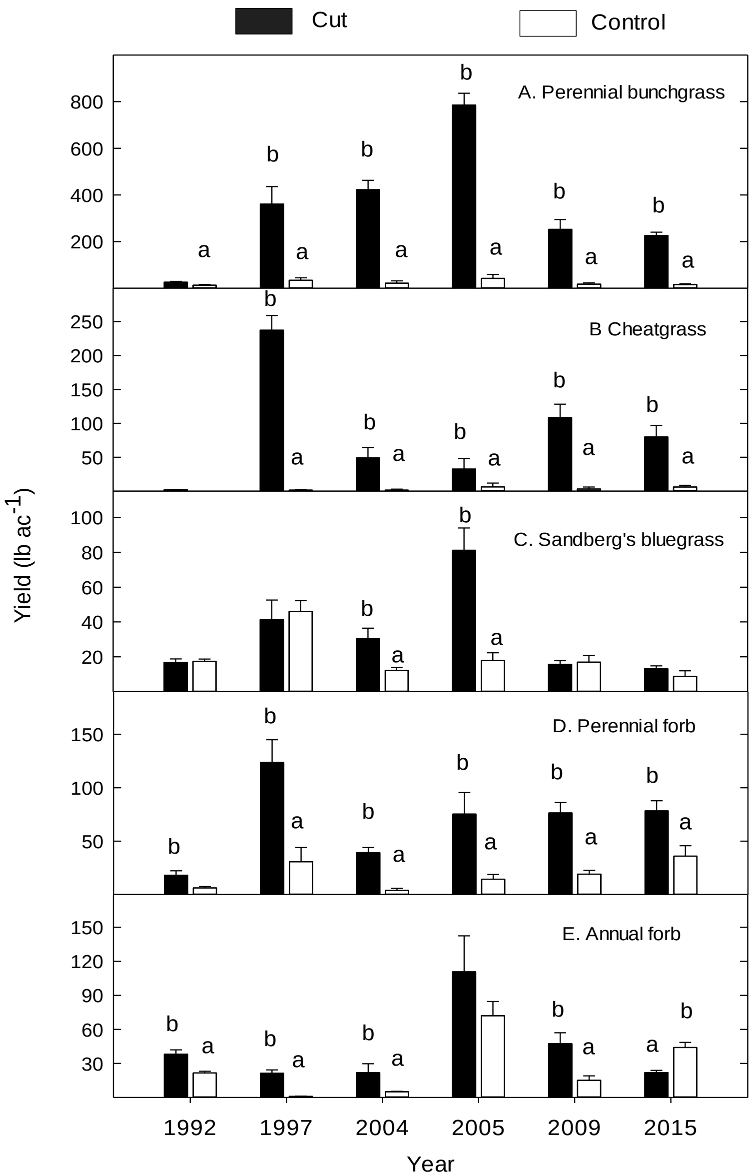

Figure 2. Herbaceous production (lb/ac), basin big sagebrush/ bluebunch wheatgrass site, Steens Mountain, 1992-2015, for A) perennial bunchgrasses, b) cheatgrass, C) Sandberg's bluegrass, D) perennial forbs, and D) annual forbs in cut and control woodlands. Cut woodland was a Phase 3. Different lower case letters indicate differences between treatments within each year for each life form.

The South Mountain sites (4A and B; Table 1) are included to demonstrate that severe fire impacts may not result in invasive annual dominance and that recovery may be incomplete over short-time horizons. Both sites were dominated by native perennial and annual forbs and perennial grass production was about 4 times greater than the controls. However, production at the two sites in 2006 was 40% to 60% less than site potential, bare ground cover was 45 to 50%, about 2 to 4 times greater than is typical of these plant communities, and perennial bunchgrass density was a third of site potentials. Although not measured since the 4th year after fire the two sites are dominated by perennial bunchgrasses and forbs, production appears to have increased by a minimum of 50% and invasive annuals are minor components of herbage production. This indicates that recovery requires patience as it takes time for new plants to establish and develop following juniper treatments, especially those that are prescribed burned.

IMPLICATIONS

Juniper treatments were all effective at increasing herbaceous production. Results indicate that fire (2 to 3-fold increase) was more effective than cutting (no change to 1.5-fold increase [2-fold possible]) at increasing herbage production in woodlands that are in early (Phase 1) and mid (Phase 2) successional stages (Table 2). Cutting Phase 1 woodlands is unlikely to increase herbage production as the presence of shrubs limits the ability of herbaceous species to respond. However, cutting, cutting/winter burning, and other mechanical conifer treatments are good preventative measures in phase 1 and 2 woodlands that at minimum, maintain herbage production as well conserving habitat characteristics for sage-grouse and other sagebrush obligate wildlife. In late successional stands (Phase 3) not containing shrubs, cutting and prescribed fire appears to result in similar increases in herbage production because of the lack of woody competition. The amount of increase in Phase 3 woodlands varies considerably (2 to 10-fold increases) depending on site potential. Cheatgrass is problematic in areas with mesic soil temperature regimes regardless of treatment and potentially may dominate areas with frigid soil temperature regimes if perennial grass density falls below 1 plant yd^{-2}. This compilation of studies suggests that site potential and vegetation starting point controls treatment responses.

Table 1. Western juniper control studies providing site, plant community, woodland phase, woodland treatment method and replication, year post treatment, fire severity, and production data (lifeform and total). Production data are means ± standard error (lbs/acre) and the values are for the last year of measurement. Different lower case letters indicate treatment differences within study site.

Study site and plant community	Woodland Phase	Years post treatment	Fire Severity	Treatments	Perennial bunchgrass	Sandberg bluegrass	Perennial forb	Annual forb	Annual grass	Total herbaceous
1. Hart Mt										
Mountain big sagebrush	Phase 1	7	Light to mod.	Rx fire[1] (5 rep)	**809 ± 90b**	**14 ± 4b**	**295 ± 31b**	19 ± 7	**145 ± 35b**	**1283 ± 118b**
grassland		(2008-2015)		Control	341 ± 82a	4 ± 2a	200 ± 27a	10 ± 3	20 ± 13a	575 ± 107a
2. NGBER[2]		4	High	Rx fire (5	**608 ± 81c**	27 ± 4	**175 ± 29b**	**32 ± 10b**	**10 ± 3b**	**852 ± 145c**
A. Mountain big sagebrush	Phase 2	(2012-2015)	Light	reps)	375 ± 17b	24 ± 2	**170 ± 45b**	8 ± 4a	2 ± 2a	584 ± 82b
/Idaho fescue				Cut[3] & burn	184 ± 33a	24 ± 3	130 ± 35a	3 ± 2a	1 ± 1a	342 ± 14a
				Control						
B. Wyoming and Mountain	Phase 1 & 2	4	High to light	Rx fire (9	**505 ± 20b**	60 ± 16	85 ± 18	**45 ± 20b**	**10 ± 6b**	**705 ± 35b**
big sagebrush grassland		(2012-2015)		reps)	190 ± 18a	55 ± 11	60 ± 20	9 ± 4a	1 ± 1a	315 ± 27a
				Control						
3. Steens Mt										
A. Mountain big sagebrush	Phase 3	8	moderate	Rx fire (5	**843 ± 55b**	97 ± 19	**328 ± 40b**	**10 + 2b**	**8 + 6b**	**1286 ± 45b**
/Idaho fescue		(2006-2012)		reps)	235 ± 12a	83 ± 10	154 ± 14a	3 ± 2a	1 ± 1a	476 ± 33a
				Control						
B. Basin big sagebrush	Phase 3	8	high		**506 ± 265**b	31 ± 7	164 ± 22	50 ± 26	**307 ± 99b**	**1058 ± 114**a
/bluebunch wheatgrass		(2006-2012)		Rx fire (4	50 ± 16a	20 ± 3	143 ± 24	39 ± 4	10 ± 5a	252 ± 27b
				reps)						
C. Basin big sagebrush	Phase 2	4	high	Control	**555 ± 42c**	23 ± 4	**137 ± 21b**	**100 ± 17b**	375 ± 63b	**1190 ± 64b**
/bluebunch wheatgrass		2012-2015	low		214 ± 39b	26 ± 6	**162 ± 11b**	**99 ± 43b**	**765 ± 69c**	**1266 ± 83b**
				Rx fire (4	65 ± 11a	24 ± 6	67 ± 13a	44 ± 5a	110 ± 29a	310 ± 41a
				reps)						
D. Basin big sagebrush	Phase 3	24	low	Cut[3] & burn	**229 ± 29b**	9 ± 2	**78 ± 9b**	22 ± 2a	**80 ± 16b**	**418 ± 23b**
/bluebunch wheatgrass		(1991-2015)		Control	40 ± 27a	13 ± 2	35 ± 9a	**44 ± 4b**	6 ± 2a	138 ± 12a
				Cut (8 reps)						
				Control						
4. South Mountain, Idaho										
A. Mountain big sagebrush	Phase 3	4	high	Rx Fire	**284 ± 37b**	15 ± 7	**193 ± 29b**	**154 ± 28b**	**65 ±**	**712 ± 43b**
/Letterman's needlegrass		(2003-2006)		Control	74 ± 20a	10 ± 5	102 ± 27a	11 ± 4a	**24b**	201 ± 38a
									4 ± 2a	
B. Mountain big sagebrush	Phase 3	4	high	Rx Fire	**275 ± 77b**	7 ± 6	**341 ± 63b**	**358 ± 99b**		**982 ± 119b**
/Columbia needlegrass		(2003-2006)		Control	72 ± 10a	10 ± 5	241 ± 19a	13 ± 5a	1 ± 1	336 ± 36a
									0 ± 0	

1 Rx (prescribed) fires were done in the fall, September and October.
2 NGBER; Northern Great Basin Experimental Range
3 clear-cutting treatments done with the use of chainsaws; cut trees were left in place or burned in winter in place.

Table 2. Summary of herbaceous production response to prescribed fire and cutting treatments in western juniper woodlands by phase in SE Oregon and SW Idaho.

	Phase 1		Phase 2		Phase 3	
Treatment	**Fire[1]**	**Cutting**	**Fire[1]**	**Cutting**	**Fire**	**Cutting**
Herbaceous production	2-3 fold increase	No change[2]	2-3 fold increase	1.5-1.8 fold increase[3]	2-10 fold increase	2-10 fold increase[4]

1 fire removes all shrubs and trees.
2 cutting only removes small trees therefore no detectable increase in herb production.
3 smaller increase than fire because of presence of shrubs
4 herb production is similar between fire and cutting as all trees removed and there is a lack of shrubs. Herb composition may differ substantially among treatments.

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