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Shrub-Steppe Early Succession Following Juniper Cutting and Prescribed Fire

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Abstract *Pinus-Juniperus* L. (Piñon-juniper) woodlands of the western United States have expanded in area nearly 10-fold since the late 1800's. *Juniperus occidentalis* ssp. *occidentalis* Hook. (western juniper) dominance in sagebrush steppe has several negative consequences, including reductions in herbaceous production and diversity, decreased wildlife habitat, and higher erosion and runoff potentials. Prescribed fire and mechanical tree removal are the main methods used to control *J. occidentalis* and restore sagebrush steppe. However, mature woodlands become difficult to prescribe burn because of the lack of understory fuels. We evaluated partial cutting of the woodlands (cutting 25–50% of the trees) to increase surface fuels, followed by prescribed fire treatments in late successional *J. occidentalis* woodlands of southwest Idaho to assess understory recovery. The study was conducted in two different plant associations and evaluated what percentage of the woodland required preparatory cutting to eliminate remaining *J. occidentalis* by prescribed fire, determined the impacts of fire to understory species, and examined early post-fire successional dynamics. The study demonstrated that late successional *J. occidentalis* woodlands can be burned after pre-cutting only a portion of the trees. Early succession in the cut-and-burn treatments were dominated by native annual and perennial forbs, in part due to high mortality of perennial bunchgrasses. By the third year after fire the number of establishing perennial grass seedlings indicated that both associations would achieve full herbaceous recovery. Cutting-prescribed fire combinations are an effective means for controlling encroaching late successional *J. occidentalis* and restoring herbaceous plant communities. However, land managers should recognize that there are potential problems associated with cutting-prescribed fire applications when invasive weeds are present.

Keywords *Bunchgrass* · *Cheatgrass* · *Juniperus occidentalis* · Mountain big sagebrush · Secondary succession · Western snowberry

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Introduction

During the past century, woodlands in many areas of the world have undergone rapid and substantial modification as a result of land use demands, alteration of historic fire regimes, invasive species, herbivore impacts, and climate changes. Ecological concerns in several woodland environments involve their loss or degradation, with examples found in South America (Bucher and Huszar 1999; Fuentes and others 1989), Asia (Ciesla and others 1998; Ciesla 2002), Australia (Hobbs and Yates 2000), and Africa (Ciesla and others 1995; Zerihun and Backleus 1991). In these regions, efforts have been focused on restoration of woodland and savanna systems (Bucher and Huszar 1999; Ciesla and others 1998; Yates and Hobbs 1997; Zerihun and Backleus 1991). In North America, Australia, and South Africa, deciduous and coniferous woodlands have expanded into shrublands (Holmes and Cowling 1997; Miller and Rose 1995; Miller and Tausch 2001; Miller and others 2005) and grasslands (Ansley and others 2001;

Archer 1994; Burrows and others 1990; Van Auken 2000), as well as altering savanna systems as tree densities increased (Peterson and others 2007). In these ecosystems, removing or reducing encroaching woodlands is a major focus of restoration efforts to maintain shrublands and grasslands.

Pinus-Juniperus L. (Piñon-juniper) woodlands of the western United States have expanded rapidly since settlement began in the late 1800's (Miller and Tausch 2001). *Juniperus occidentalis* ssp. *occidentalis* Hook. (western juniper) woodlands found in the northern Great Basin and Columbia Plateau and have increased from about 0.3 million ha to nearly 3.5 million ha during the past 130 years in eastern Oregon, southwestern Idaho, and along the northern border of California and Nevada (Miller and others 2000). The main cause of the expansion has been attributed to reductions in fire disturbance as a consequence of grazing induced fine fuel reduction and fire suppression (Burkhardt and Tisdale 1969; Miller and Rose 1995). Prior to Euro-American settlement, mean fire return intervals (MFRI) in sagebrush steppe, sufficient to prevent *J. occidentalis* woodland development, have been estimated at between10 and 50 years (Burkhardt and Tisdale 1976; Miller and Rose 1995; Miller and others 2005; Miller and Heyerdahl 2008; Wright and Bailey 1982).

Encroaching *J. occidentalis* woodlands have been categorized into three successional phases (Miller and others 2005). Phase 1 woodlands occur when shrubs and herbaceous species are the dominant vegetation component with few trees present. In Phase 2 woodlands, *J. occidentalis* co-dominates with shrub and herbaceous layers. Phase 3 woodlands occur when *J. occidentalis* is dominant and is the primary component influencing ecological processes. The negative impacts of *J. occidentalis* invasion include increased soil erosion (Buckhouse and Mattison 1980; Pierson and others 2007), loss of wildlife habitat (Noson and others 2006; Reinkensmeyer and others 2007; Schaefer and others 2003), and reduced plant community diversity and herbaceous productivity (Bates and others 2000; Bates and others 2005; Bates and others 2006; Miller and others 2000; Miller and others 2005).

Prescribed fire, tree cutting, and a combination of these treatments have been the main methods used to control *J. occidentalis* and restore sagebrush steppe, *Populus tremuloides*(quaking aspen) woodlands, and riparian communities (Miller and others 2005). Harvesting of *J. occidentalis* as a bio-fuel for energy production has largely been confined to northeastern California. Elsewhere, commercial use of cut trees has not been economically viable because of transportation costs, inadequate infrastructure, cheaper energy alternatives, and concerns of long-term supply availability.

Fire remains a viable management option for *J. occidentalis* control in woodlands that are in early (Phase 1) to mid successional (Phase 2) stages when sufficient and continuous surface (0–1 m) fuels are present (Miller and others 2005). In late successional (Phase 3) woodlands, surface fuels are typically not adequate to sustain fire and kill trees. Clear-cutting and either removing or retaining cut trees on the ground remains the most common practice for managing late successional woodlands. Cutting treatments using chainsaws and tractor mounted shears has been successful at recovery of shrub-steppe plant communities (Rose and Eddleman 1994; Bates and others 2005; Bates and others 2007a; Miller and others 2005). Cutting and prescribed fire combinations are relatively recent methods for treating Phase 2 and Phase 3 *J. occidentalis* woodlands and have been applied extensively in eastern Oregon and northern California the past decade. These methods entail cutting about 30% of the trees to increase surface fuel levels to carry fire and kill remaining trees (Bates and others 2005; Bates and others 2007b; Miller and others 2005).

We evaluated partial cutting and prescribed fire treatments in Phase 3 *J. occidentalis* woodlands. Two levels of preparatory *J. occidentalis* cutting followed by prescribed fall fire were evaluated. The objectives of the study were to: (1) assess what percentage of woodlands required preparatory cutting to eliminate remaining *J. occidentalis* trees by prescribed fire; (2) determine fire impacts to understory species; and (3) evaluate early post-fire successional dynamics. We hypothesized the fire portion of the treatment would severely impact herbaceous understory, particularly bunchgrasses, by causing high mortality. A previous study indicated that fall burning of *P. tremuloides* woodland after partial cutting (one-third of trees cut) of encroaching *J. occidentalis* (Phase 3 woodland) eliminated the majority of perennial bunchgrasses and suppressed perennial forbs (Bates and others 2007b). Based on successional models developed for post-fire *Pinus-Juniperus* woodlands (Barney and Frischknecht 1974), we hypothesized that early succession would be characterized by high cover and abundance of annual and perennial forbs, with the potential for invasive grasses to increase because *Bromus tectorum* L. (cheatgrass) and *Poa bulbosa* L. (bulbous bluegrass) were present.

Methods

Study Area

The study was near South Mountain, Idaho, about 115 km south-southwest of Boise. Two plant associations were selected for treatment and were designated as Columbiana and Needlegrass. The Columbiana association occurred on north facing aspects, and the Needlegrass association occurred on west and southwest facing aspects. The Columbiana association was characterized by *Artemisia tridentata* Nutt. spp. *vaseyana* (Rydb.) Beetle-*Symphoricarpus oreophilus* Gray/*Achnatherum nelsonii* (Scribn.) Barkworth-*Festuca idahoensis* Elmer (mountain big sagebrush-mountain snowberry/Columbia needlegrass-Idaho fescue) plant communities. The Needlegrass association was comprised of *Artemisia tridentata* spp. *vaseyana*/*Achnatherum lettermanii* (Vasey) Barkworth-*Pseudoroegenaria spicata* (Pursh) A. Löve (Mountain big sagebrush/Letterman's needlegrass-bluebunch wheatgrass) plant communities. Both are representative of plant associations found between 1,525 and 1,800 m that are being invaded by western juniper in southwest Idaho. Elevation at the study sites was about 1,650 m. Both plant associations were dominated by post Euro-American settlement (<130 year old) *J. occidentalis* woodlands (Phase 3 woodlands). *J. occidentalis* encroachment had largely eliminated the shrub layer and depleted the understory. *Poa bulbosa*, a nonnative invasive perennial grass, was the most common herbaceous plant in both associations. *Bromus tectorum*, a nonnative annual grass, was present in trace amounts on the Needlegrass association.

Climate is typical of the northern Great Basin, with the majority of precipitation arriving between November and May, whereas summers are warm and dry. Annual precipitation (October 1–Sept. 30) at the SNOTEL site on South Mountain (12.5 km north of the study area at 1,980 m elevation) has averaged 853 mm the past 25 years (1982–2006) (Natural Resource Conservation Service [NRCS] 2009). Precipitation at the South Mountain SNOTEL, which is 330 m higher in elevation, is likely greater than our study sites, as western juniper typically grows in areas receiving 260–460 mm of annual precipitation (Miller and others 2005). Ecological site descriptions for the study's associations indicate that they were in a 305–406 mm precipitation zone (Natural Resource Conservation Service [NRCS] 2010a). Precipitation was about average during the post-fire (2003–2006) period. Soils were described to the subgroup at each site and were identified as Pachic Argixerolls. The Columbiana sites had a deeper A horizon (A1, 0–10 cm; A2, 10–30 cm) than the Needlegrass sites (A, 0–6 cm). Soil pH of the A horizon was 6.7 on the Columbiana sites and 7.4 in the Needlegrass sites.

Experimental Design and Treatment Application

The experimental design for each association was a randomized complete block design (Peterson 1985). Preparatory tree manipulations involved cutting trees with chainsaws in October 2002. Two levels of cutting were applied and included cutting 25% and 50% of post-settlement trees (based on tree density). For the Needlegrass association, treatments were Needlegrass25 (25% of the trees cut) and Needlegrass50 (50% of the trees cut). Similarly, for the Columbiana association, treatments were Columbiana25 and Columbiana50. Uncut and unburned woodlands (Needlegrass Control, Columbiana Control) were located adjacent to treated plots. Each treatment plot was 1.0 hectare in size and was replicated 5 times (40 plots total) per association. Cut trees dried for one year prior to fire application in fall 2003. Prescribed fires (strip head fire technique) were applied on October 21–22, 2003. All plots, aside from one Needlgrass25 plot, were successfully burned. A control plot in the Columbiana association was lost as a result of fire over-run. Burn conditions were typical for prescribed fire applications for treating encroaching western juniper woodlands (Table 1).

Fire severity was estimated by applying a severity index used by Bates and others (2006) for evaluating *P. tremuloides* community response to fire. The severity categories were light (1–30% mortality of perennial bunchgrasses, needles and small branches of downed *J. occidentalis* consumed, and ≤20% of western juniper killed), moderate (31–70% mortality of perennial bunchgrasses, large branches and trunks remained on downed *J. occidentalis*, and <70% of juniper killed), and high (71–100% mortality of perennial bunchgrasses, only trunks of downed *J. occidentalis* remaining, and >90% of *J. occidentalis* killed).

Table 1 Weather, fuel moisture, and fire conditions for western juniper cutting -prescribed fire treatments in mountain big sagebrush communities, South Mountain, Idaho in October, 2003

	Plant community	
	Needlegrass	Columbiana
Atmospheric conditions		
Air temperature (°C)	15–24	15–24
Relative humidity (%)	18–23	18–23
Wind speed (kph)	5–16	5–16
Soil water content (%, 0–10 cm)	7.6 ± 0.8	9.6 ± 0.5
Fuel moisture (%)		
Herbaceous	4.6 ± 0.2	5.7 ± 0.1
Surface litters	5.4 ± 0.4	5.9 ± 0.2
10 h	5.2 ± 0.4	6.0 ± 0.4
100 h	7.0 ± 0.2	7.3 ± 0.5
1000 h	10.8 ± 0.7	10.5 ± 0.6
Fire conditions		
Soil temp (2 cm below surface)		
Interspace/woodland floor (°C)	<79	79–204
Beneath cut trees (°C)	630–816	704–816
Canopy litter mats (°C)	177–630	204–704
Burn duration (min)	5–43	5–55
Flame length (m)	2–10	2–11

Perennial bunchgrass mortality estimates were derived from pre- and first year post-fire measurements of bunchgrass density.

Gravimetric soil water (0–10 cm) and fuel moisture for herbaceous fine fuels, litter, 1, 10, 100, and 1000-hour fuels were measured the day of fire application (Table 1). Fuel moisture and soil water content were determined by drying samples at 100°C to a constant weight. Weather data (RH, wind speed, temperature) were recorded prior to and during fire applications. Temperature and relative humidity were typical for fall prescribed fire application in the region. Burn duration (active flame) and flame lengths were also estimated.

Soil temperatures during the fires were estimated using Tempilaq[1] paints applied to 25 × 80 × 0.4 mm steel tags. Tempilaq paints melt or discolor at specific temperatures when heat is applied. Five sets of tags were placed 1 cm below the soil surface in each plant community at three locations. Tags were placed in interspaces, beneath live trees, and beneath cut trees. Sets consisted of 20 individual indicator tags and each tag was marked with its own indicator paint. Twenty temperature paints were used from 79°C to 1093°C (intervals between temperatures varied from 14°C at the lower temperatures to about 56°C at the higher temperatures). Temperature values were etched on the metal tags for identification.

Measurements

Pre-treatment vegetation (trees, shrubs, herbaceous) measurements were collected in June 2002. Post-fire measurements were gathered in June, 2004–2006. On each treatment plot, four 50-m transects were permanently established, with transects spaced 20 m apart. Cover of *J. occidentalis* and shrubs were estimated by line intercept (Canfield 1941) along each transect. Density of mature *J. occidentalis* (>2 m height) was estimated by counting all rooted individuals along four, 6 × 50 m belt transects. Density of shrubs and juvenile *J. occidentalis* (<2 m height) were estimated by counting all rooted individuals along four, 2 × 50 m belt transects. Understory canopy cover (by species) and herbaceous perennial density (by species) was sampled inside 0.2 m^2 frames (0.4 × 0.5 m). Frames were placed every 2 m along transect lines. A species list (richness) was compiled for each treatment plot, with scientific nomenclature following the Natural Resource Conservation Service Plant Database (Natural Resource Conservation Service [NRCS] 2010b) and Hitchcock and Cronquist (1987). Herbaceous production was measured by functional group in 2006, using 15, 1-m^2 frames per treatment plot. Herbage was clipped to 4-cm stubble for perennial bunch grasses and to ground level for rhizomatous perennial grasses, annual grasses, and perennial and annual forbs.

Statistical Analysis

Repeated measures analysis of variance (PROC MIXED procedure, SAS Institute, Cary, North Carolina) for a randomized complete block design was used to test for year, treatment, and year by treatment interaction for herbaceous, shrub, and *J. occidentalis* response variables. Plant associations were analyzed separately because of differences in herbaceous composition and soil characteristics. Response variables were *J. occidentalis* cover and density, shrub cover and density, cover (species and life form, bare ground, and surface litter), and herbaceous density (species and life form). Herbaceous life forms were grouped as *Poa secunda* Vasey (Sandberg's bluegrass), *P. bulbosa*, rhizomatous perennial grasses (primarily *Poa pratensis* L. (Kentucky bluegrass)), deep-rooted perennial bunchgrasses (e.g., Idaho fescue, Columbia needlegrass, and Letterman's needlegrass), *B. tectorum*, perennial forbs, and annual forbs. An auto regressive order one covariance structure was used because it provided the best fit for data analysis (Littell and others 1996). Mean separation involved comparison of least squares using the LSMEANS statement (SAS Institute 2002). The models included block (5 blocks; $df = 4$), year ($df = 3$), treatment ($df = 2$), and year by treatment interaction ($df = 6$; with the error term $df = 92$). Because of a strong year effect, years were analyzed separately using a general linearized model (PROC GLM, SAS Institute, 2007) for a randomized complete block to simplify presentation of results and to assist in explaining interactions (model: 5 blocks, $df = 4$; 4 treatments, $df = 3$). Data were tested for normality using the SAS univariate procedure. Data not normally distributed were arcsine square-root transformed to stabilize variance. Back transformed means are reported. Statistical significance for all tests was set at $P < 0.05$.

Results

Juniper Removal and Fire Severity

The burn treatments in both associations eliminated almost all remaining *J. occidentalis* trees, regardless of cutting level. On the Columbiana association, the fires killed 95–99% of the remaining trees, and *J. occidentalis* cover was reduced by 99% (Fig. 1a, b). On the Needlegrass

[1] Tempilaq paints are manufactured by Tempil, South Plainfield, New Jersey, 07080, USA. Mention of trade names does not imply endorsement by USDA-ARS, Eastern Oregon Agricultural Research Center, and Oregon State University.

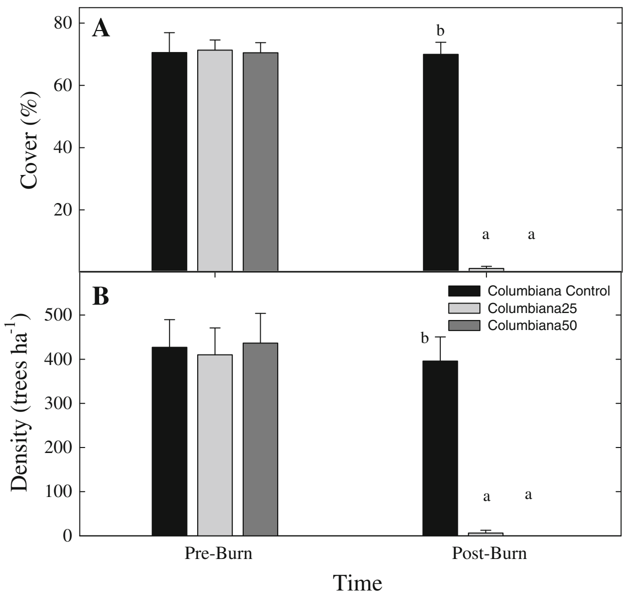

Fig. 1 Pre-fire and post-fire western juniper cover (**a**) and density (**b**) for the Columbiana plant association, South Mountain, Idaho (2002–2006). Treatments are Columbiana Control (untreated control), Columbiana 25 (25% cut and prescribed burn), and Columbiana 50 (50% cut and prescribed burn). Data are in means ± SE. Means sharing a common lower case letter are not significantly different ($P > 0.05$)

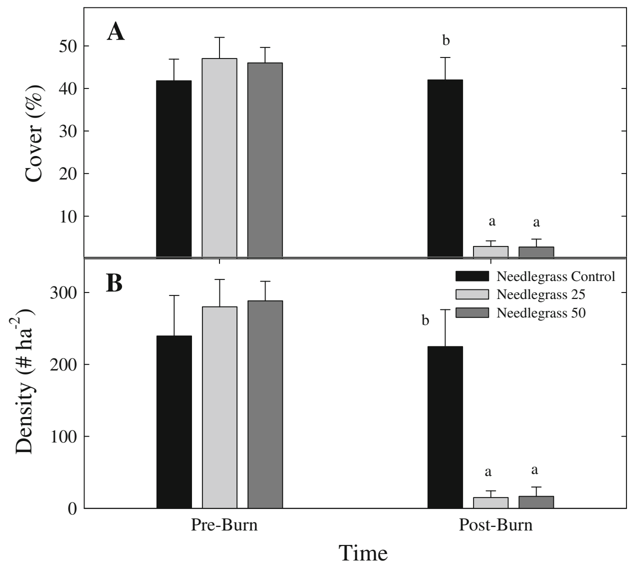

Fig. 2 Pre-fire and post-fire western juniper cover (**a**) and density (**b**) for the Needlegrass plant association, South Mountain, Idaho (2002–2006). Data are in means ± SE. Means sharing a common lower case letter are not significantly different ($P > 0.05$)

association, fires killed 90–100% of the remaining trees, and *J. occidentalis* cover was reduced by 90–100% (Fig. 2a, b). In both plant associations, cut trees were either fully consumed or only the trunks remained. Perennial bunchgrass mortality exceeded 80% on the Columbiana plant association and all surface litter was consumed. Fires on the Columbiana association were judged to have been of high severity. Although litter and fuel consumption were largely complete in the Needlegrass association, mortality of perennial bunchgrasses was about 50%, thus, fire severity was concluded to be moderate. Soil temperatures during the fire were greater beneath cut trees and in canopy litter mats around stumps or live trees than the interspace and woodland floor. In the Needlegrass association, interspaces between trees were primarily bare ground, and soil temperature did not exceed 79°C. In both associations, soil temperatures (2 cm depth) beneath cut trees exceeded 704°C.

Columbiana Association: Herbaceous and Shrub Dynamics

Cover of perennial bunchgrasses ($P = 0.839$) and perennial forbs ($P = 0.958$) did not differ among the treatments, although cover increased by 100–130% and 100–180% in 2005 and 2006, respectively (Fig. 3a, b; $P < 0.0001$). Perennial forb species that had greater cover in the Columbiana25 and Columbiana50 treatments after fire than the control were *Astragalus lentiginosus* Dougl. (speckle-pod milkvetch) (15% greater; $P = 0.040$), *Lupinus arbustus* L. (spur lupine) (20% greater; $P = 0.019$), and *Hydrophyllum capitatum* Dougl. (ballhead waterleaf) (10% greater; $P = 0.005$). Cover of *P. bulbosa* was reduced by about 70% in both cut-burn treatments (Fig. 3c; $P < 0.001$). Annual forb cover was 4–10-fold greater in the cut-and-burn treatments in the second and third year after fire (Fig. 3d; $P < 0.0001$). Total herbaceous cover was 70–100% greater in both cut-and-burn treatments in the second and third year after fire (Fig. 3e; $P < 0.001$). Annual species that had greater cover in both cut-burn treatments than the Columbiana control were *Collinsia parviflora* (Lindl) (blue eyed Mary) (2–7 fold greater; $P = 0.042$), *Collomia linearis* Nutt. (narrow-leaf collomia) (10–150 fold greater; $P = 0.001$), *Claytonia perfoliata.* (miner's lettuce) (2–15 fold greater; $P = 0.001$), *Epilobium minutum* (Lindl. ex Lehm) (willow-weed) (2–15 fold greater; $P = 0.017$), and *Cryptantha* spp. (Lehm.) (>1000 fold greater; $P < 0.001$). Other species and herbaceous functional groups exhibited neither treatment differences nor changes over time. Bare ground was 80–300% greater in the treated plots than the Columbiana control after fire ($P < 0.001$; Fig. 4a). The increase in bare ground resulted from a >90% reduction of *J. occidentalis* litter in the cut-burn treatments ($P < 0.001$; Fig. 4b). Herbaceous litter was 2–2.5 fold higher in the cut-burn treatments than the Columbiana control in 2006 ($P < 0.001$; Fig. 4c).

Perennial bunchgrass density was reduced by 70–85% after fire, thus, the Columbiana25 and Columbiana50

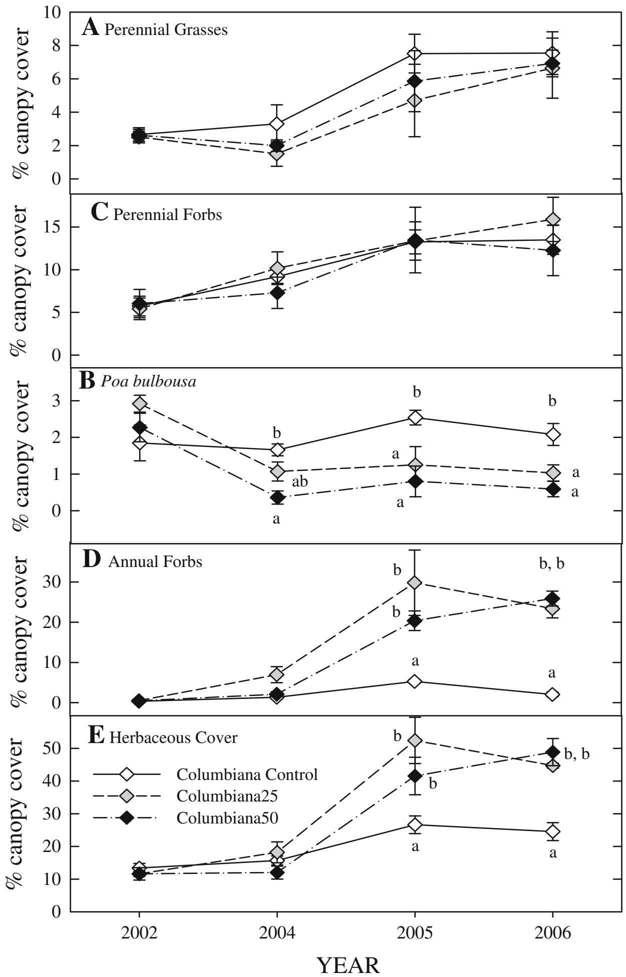

Fig. 3 Functional group cover (%) for the Columbiana plant association, South Mountain, Idaho (2002–2006): **a** perennial grasses; **b** bluegrass spp. (*P. bulbosa* and *P. secunda*); **c** perennial forbs; **d** annual forbs; and (**e**) total herbaceous. Treatments are Columbiana Control (untreated control), Columbiana 25 (25% cut and prescribed burn), and Columbiana 50 (50% cut and prescribed burn). Data are in means ± SE. Means sharing a common lower case letter are not significantly different ($P > 0.05$)

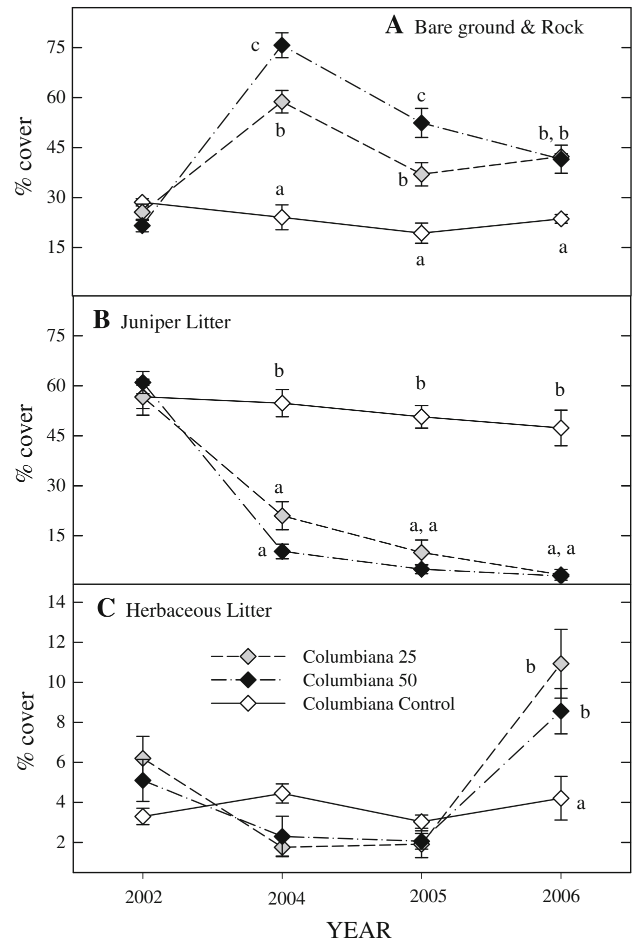

Fig. 4 Bare ground and litter covers (%) for the Columbiana plant association, South Mountain, Idaho (2002–2006): **a** bare ground and rock; **b** juniper litter; and **c** herbaceous litter. Treatments are Columbiana Control (untreated control), Columbiana 25 (25% cut and prescribed burn), and Columbiana 50 (50% cut and prescribed burn). Data are in means ± SE. Means sharing a common lower case letter are not significantly different ($P > 0.05$)

treatments were less than the Columbiana control ($P = 0.002$; Fig. 5a). In the third year after fire, bunchgrass grass seedling density was 100–180-fold greater in the cut-and-burn treatments indicating that perennial grasses were recovering rapidly ($P < 0.0001$; Fig. 5b). Perennial forb density was 2-fold greater in the Columbiana control than the Columbiana25 treatment in 2006 ($P = 0.003$; Fig. 5c). *Poa bulbosa* density was reduced by 95% in both cut-and-burn treatments after fire ($P = 0.008$). The number of perennial forb ($P = 0.027$), annual forb ($P < 0.001$), and total herbaceous ($P < 0.001$) species in all treatments increased by 40–60% between 2002 and other measurement years but did not differ among treatments (perennial forb, $P = 0.078$, annual forb, $P = 0.265$, total herbaceous, $P = 0.251$).

Total herbaceous production was about 3-fold greater in the Columbiana25 and Columbiana50 treatments than the Columbiana control in 2006 ($P = 0.020$; Fig. 6). Annual forb biomass was the only functional group that was greater in the cut-burn treatments than the Columbiana control (15–28 fold greater; $P = 0.015$).

A. tridentata spp. *vaseyana* was eliminated by the burn treatments. However, because densities and cover values were low prior to burning, treatments did not differ ($P = 0.526$ and $P = 0.456$, respectively). *S. oreophilus* and *Chrysothamnus nauseous* (Pall) Britt. (gray rabbitbrush) resprouted after fire, however, no treatment differences were detected in density and cover. *Ceanothus*

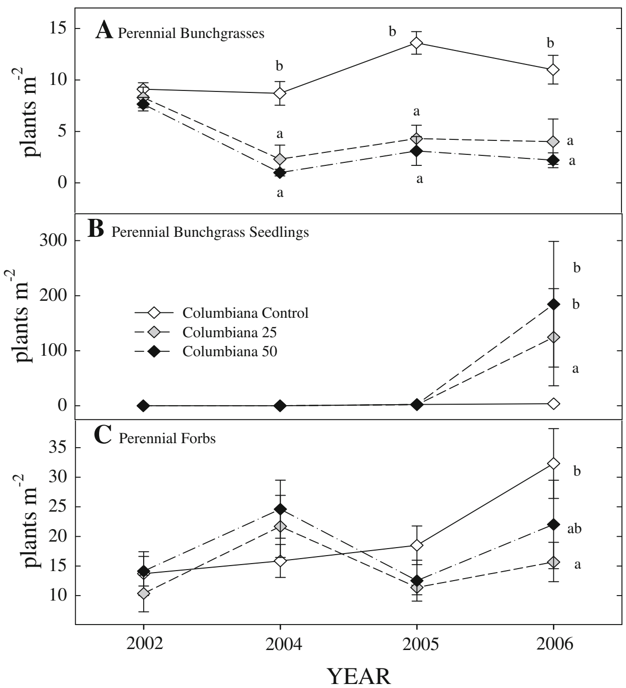

Fig. 5 Herbaceous perennial densities (# m^{-2}) in the Columbiana plant association, South Mountain, Idaho (2002–2006): **a** perennial grasses; **b** perennial forbs; **c** perennial bunchgrass seedlings. Treatments are Columbiana Control (untreated control), Columbiana 25 (25% cut and prescribed burn), and Columbiana 50 (50% cut and prescribed burn). Data are in means ± SE. Means sharing a common lower case letter are not significantly different ($P > 0.05$)

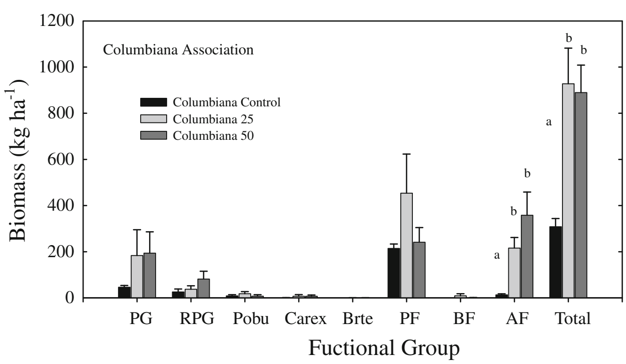

Fig. 6 Biomass (kg ha^{-1}) production in the Columbiana plant association, South mountain, Idaho, 2006. Functional groups are; perennial bunchgrass (PG); rhizomatous grasses (RPG); *P. bulbosa* (Pobu); carex species (Carex); *B. tectorum* (Brte); native perennial forb (PF); non-native biennial forb (BF); annual Forb (AF); and total herbaceous production. Treatments are Columbiana Control (untreated control), Columbiana 25 (25% cut and prescribed burn), and Columbiana 50 (50% cut and prescribed burn). Data are in means ± SE. Means sharing a common lower case letter are not significantly different ($P > 0.05$)

velutinus Dougl. (snowbrush), which was not present prior to burning, increased in density in the Columbiana25 and Columbiana50 treatments ($P = 0.0056$).

Needlegrass Association: Herbaceous and Shrub Dynamics

Perennial bunchgrass cover increased about 3 fold in all treatments during the study ($P < 0.001$; Fig. 7a). *Bromus tectorum* cover did not change in response to treatment or across years ($P = 0.082$ and $P = 0.235$, respectively; Fig. 7b). *P. bulbosa* was reduced in cover by 80% in the Needlegrass25 and Needlgrass50 treatments ($P = 0.031$). Cover of perennial forbs was 2–3 fold higher in the cut-burn treatments than the Needlegrass control ($P = 0.018$; Fig. 7c). Annual forbs cover was 4–10-fold greater and total herbaceous cover was 2.5 times greater in the cut-burn treatments than the control in the second and third year after fire ($P < 0.001$ and $P = .005$, respectively; Fig. 7d, e). Bare ground increased about 15% the first year after fire in the cut-burn treatments, however, in subsequent years

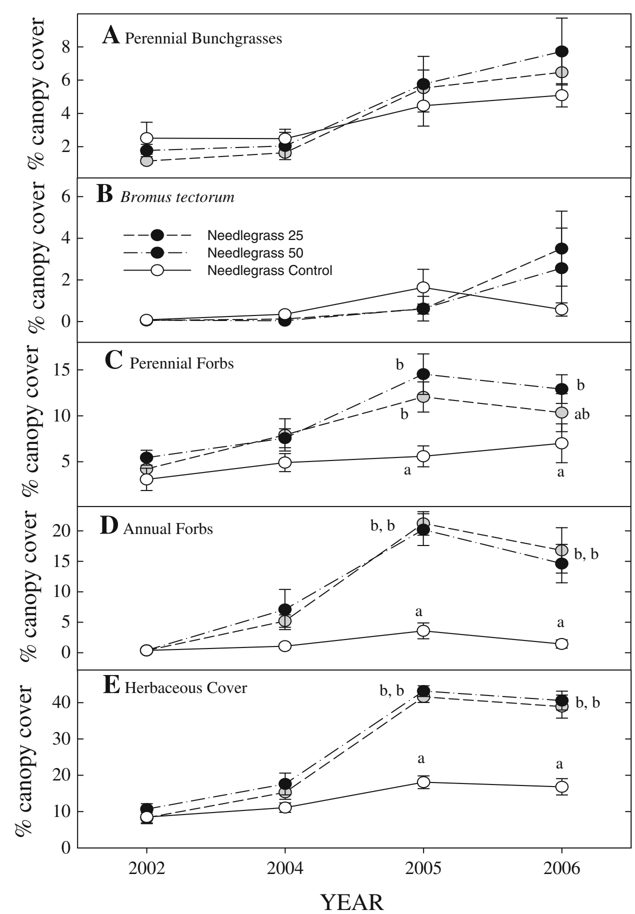

Fig. 7 Functional group cover (%) for the Needlegrass plant association, South Mountain, Idaho (2002–2006): **a** perennial grasses; **b** *B. tectorum*; **c** perennial forbs; **d** annual forbs; and **e** total herbaceous. Treatments are Needlegrass Control (untreated control), Needlegrass 25 (25% cut and prescribed burn), and Needlegrass 50 (50% cut and prescribed burn). Data are in means ± SE. Means sharing a common lower case letter are not significantly different ($P > 0.05$)

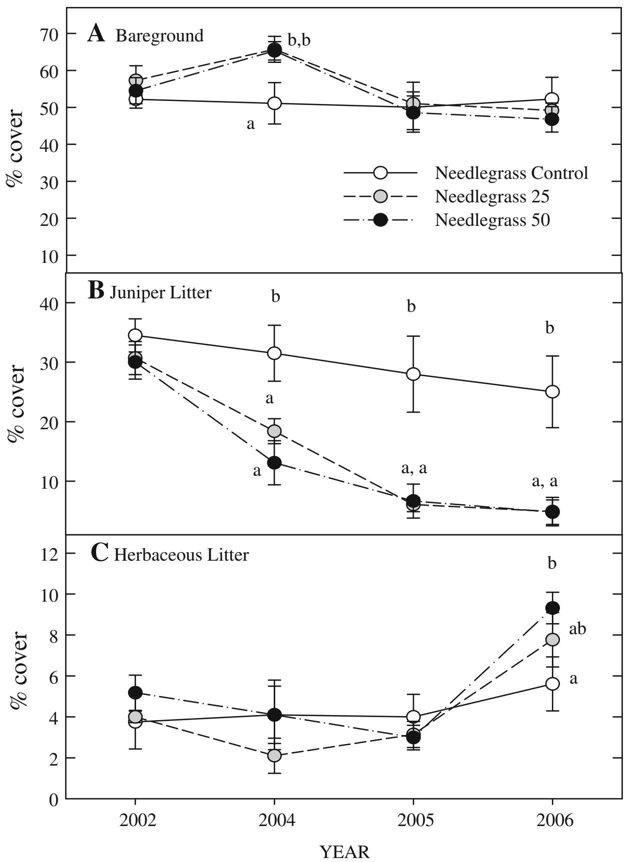

Fig. 8 Bare ground and litter covers (%) for the Needlegrass plant association, South Mountain, Idaho (2002–2006): **a** bare ground and rock; **b** juniper litter; and **c** herbaceous litter. Treatments are Needlegrass Control (untreated control), Needlegrass 25 (25% cut and prescribed burn), and Needlegrass 50 (50% cut and prescribed burn). Data are in means ± SE. Means sharing a common lower case letter are not significantly different ($P > 0.05$)

there were no treatment differences ($P = 0.308$; Fig. 8a). *J. occidentalis* litter was reduced about 5-fold after fire in both cut-burn treatments ($P = 0.045$; Fig. 8b). Herbaceous litter increased in all treatments by about 50% and was greatest in the Needlgrass50 treatment the third year after fire ($P = 0.0173$; Fig. 8c). Annual forb species that increased after fire in the cut-burn treatments and were greater than the control were *C. parviflora* (2–5 fold greater, $P = 0.002$), *C. linearis* (2–7 fold greater, $P = 0.001$), *M. gracilis* (3–4 fold greater, $P = 0.005$), *Cyrptantha* spp. (100 to140 fold greater, $P < 0.001$) and *Gayophytum* spp. (4–7 fold greater, $P < 0.001$).

Perennial bunchgrass density decreased 50% after fire ($P = 0.005$) in the Needlegrass25 and Needlgrass50 treatments, and both remained less than the Needlegrass control during the study ($P = 0.0017$; Fig 9a). However, in 2006, the third year after fire, grass seedling density was 30–60 times greater in the cut-and-burn treatments than the Needlegrass control ($P = 0.023$; Fig. 9b). Densities of *P. bulbosa* ($P = 0.031$) and *P. secunda* ($P = 0.0364$) were reduced 75% and 50% respectively in the cut-burn treatments. Perennial forb density increased in all treatments after 2002 ($P < 0.001$); however the increase was greater in the cut-burn treatments than the Needlegrass control ($P = 0.018$; Fig. 9c). The number of perennial ($P < 0.001$) and annual forb species ($P < 0.001$) identified increased by 30% in 2005 and 2006 and were greater in both cut-burn treatments than the control. *A. tridentata* spp. *vaseyana* and *Purshia tridentata* (Pursh) DC. (antelope bitterbrush) remained present in the cut-burn treatments, and there were no treatment differences ($P = 0.336$). *C. velutinus* increased in density in both treatments the first growing season after fire and was greater than the Needlegrass control ($P = 0.027$).

Production of perennial bunchgrass ($P = 0.017$), perennial forbs ($P = 0.028$), annual forbs ($P = 0.022$), and total herbaceous biomass ($P < 0.001$) was greater in the Needlegrass25 and Needlgrass50 treatments in 2006 (Fig. 10). Perennial grass biomass and total herbaceous biomass were about three times greater in the cut-burn treatments than the control.

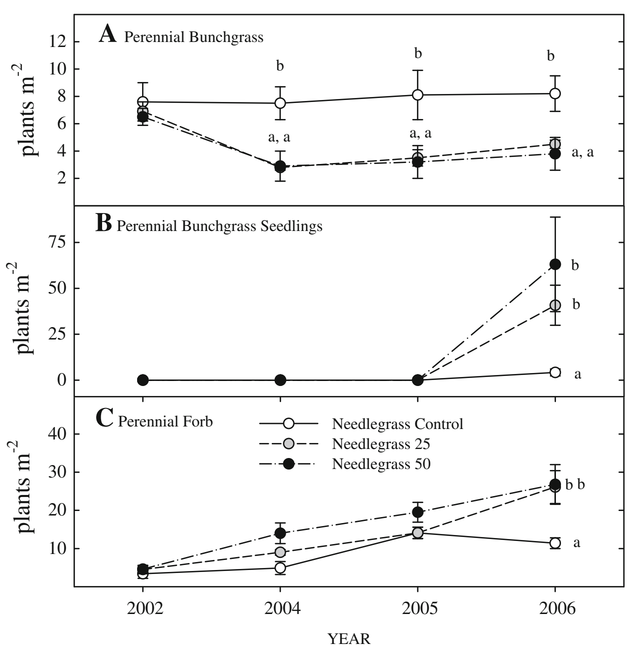

Fig. 9 Herbaceous perennial densities (# m^{-2}) for the Needlegrass plant association, South Mountain, Idaho (2002–2006): **a** perennial grasses; **b** perennial forbs; **c** perennial bunchgrass seedlings. Treatments are Needlegrass Control (untreated control), Needlegrass 25 (25% cut and prescribed burn), and Needlegrass 50 (50% cut and prescribed burn). Data are in means ± SE. Means sharing a common lower case letter are not significantly different ($P > 0.05$)

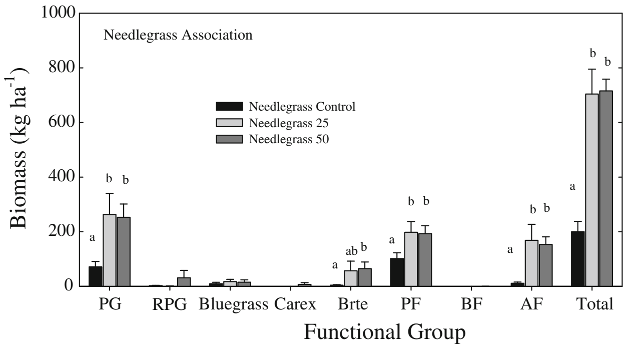

Fig. 10 Biomass (kg ha^{-1}) production in the Needlegrass plant association, South mountain, Idaho, 2006. Functional groups are; perennial bunchgrass (PG); rhizomatous grasses (RPG); *P. bulbosa* and *P. secunda* (Bluegrass); carex species (Carex); *B. tectorum* (Brte); native perennial forb (PF); non-native biennial forb (BF); annual Forb (AF); and total herbaceous production. Treatments are Needlegrass Control (untreated control), Needlegrass 25 (25% cut and prescribed burn), and Needlegrass 50 (50% cut and prescribed burn). Data are in means ± SE. Means sharing a common lower case letter are not significantly different ($P > 0.05$)

Discussion

Treatment Differences

This study demonstrated that late successional (Phase 3) *J. occidentalis* woodlands can be burned successfully after pre-cutting a portion of the trees. After drying for one year, the cut trees carried the fire, which killed most remaining *J. occidentalis*. We had expected a gradient of response in *J. occidentalis* control and understory recovery. However, the levels of *J. occidentalis* control, litter and fuel consumption, mortality of perennial bunchgrasses, and understory response were independent of cutting level because treatments within each association responded similarly to the fires.

In the Columbiana association, the lack of differences between the cut-burn treatments was likely a result of woodland characteristics, fuel conditions, and weather. The woodland floor was largely covered with a mat of *J. occidentalis* leaf litter and, relative to the Needlegrass community, the higher tree cover and density resulted in more material on the ground after cutting. Coupled with very dry conditions and high fuel consumption, the fire treatments had equally severe impacts to the understory. Perennial grass crowns were typically observed to be elevated above the soil in the *J. occidentalis* litter layer, which likely contributed to the high mortality. The impacts of the fires to the understory and early successional stages were similar to those documented for high intensity/severity fires in many forested systems of the western United States and Canada (Brown and Smith 2000). In *Pinus ponderosa* Dougl. (ponderosa pine) forest, perennial grass cover decreased as fire intensity and litter consumption increased (Armour and others 1984). In *P. tremuloides* woodland, pre-cutting one-third of encroaching *J. occidentalis* followed by prescribed fire in the fall resulted in complete consumption of the organic layer, killed almost all perennial grasses, and severely reduced perennial forb recovery (Bates and others 2006).

In the Needlegrass community, the interspace was mostly bare ground. Cutting in similar, open *J. occidentalis* stands reduces the interspace area because of increased coverage by down trees (Bates and others 1998; Bates and Svejcar 2009). Because the Needlegrass50 treatment had twice as many trees on the ground as the Needlegrass25 treatment we had expected to measure greater negative impacts of fire to understory vegetation. Although we did not differentiate among interspaces and areas occupied by cut or standing trees most of the perennial bunchgrass mortality appeared to be beneath down or standing trees, where there was an accumulation of *J. occidentalis* leaf litter. We suspect that cutting 25% of the trees compared to cutting 50% of the trees may not produce that much of a difference (less than 5%) in the amount of interspace covered by trees. This likely explains the lack of differences between the two cut-and-burn treatments in perennial grass mortality and post-fire understory response.

Western Juniper Control

Cutting a minimum of 25% of the mature trees in communities dominated by *J. occidentalis* (cover of 35–70%) was sufficient to remove the majority of remaining live trees during fall prescribed fire application. Cutting more than 25% of the trees was unnecessary when broadcast burning was applied with weather conditions typically encountered with fall prescribed fire. Burning was equally effective on slopes (Columbiana and Needlegrass sites, 10–60% slopes) and on flat ground (Columbiana sites). The 50% pre-fire cutting levels were excessive, particularly in the Columbiana plant community. In *P. tremuloides* woodland, with similar levels of cover and density of *J. occidentalis*, only a third of the trees were cut prior to prescribed fire (Bates and others 2006). Cutting less than 25% (10–20%) of the woodland trees in the Columbiana plant community would likely have been sufficient to kill remaining *J. occidentalis* under the conditions the prescribed fire was applied and may have reduced the adverse effects to the herbaceous layer. If management objectives are to retain some trees on site reducing cutting levels may also reduce fire mortality of remaining *J. occidentalis,* especially in the Needlegrass association which was a more open stand.

The plot size used in our study was small (1 ha), however, the treatments applied are readily scaled up to larger landscape projects in other *Pinus-Juniperus* woodlands. Large scale projects (300–2500 ha) conducted by federal land management agencies, involving the cutting of about 33% of the trees prior to fire, have been used in several shrub-steppe restoration studies in *J. occidentalis* woodlands (Bates and others 2006; Bates and others 2007b). Approximately 30,000 ha of *J. occidentalis* woodlands have been treated using partial cutting and prescribed fire control measures in eastern Oregon. At the larger scales, managers can adjust cutting levels or leave some woodland areas uncut to create a landscape mosaic of burned and unburned areas.

Understory Release

Secondary succession in woodland and forest ecosystems after wild or prescribed fire often varies in composition and recovery rate as a result of varying fire severity, nutrient and water availability, site characteristics, herbivory, and weather (Armour and others 1984; Barney and Frischknecht 1974; Griffis and others 2001; Koniak 1985; Oswald and Covington 1983; Sabo and others 2009). Because of severe fire effects to perennial grasses, early succession in the cut-and-burn treatments in both plant communities was largely dominated by annual and perennial forbs. These results are similar to early successional dynamics following wildfire in *Pinus-Juniperus* woodlands of Nevada (Barney and Frischknecht 1974; Koniak 1985), and ponderosa pine forests in Arizona (Griffis and others 2001; Laughlin and Fulé 2008; Sabo and others 2009). The increase in understory productivity and cover following overstory removal in *J. occidentalis* woodlands has been linked to increased soil water and nitrogen availability (Bates and others 2000; Bates and others 2002).

Both Columbiana and Needlegrass sites remain open to colonization by native and invasive species. However, because there were large numbers of perennial bunchgrass seedlings in 2006, both communities had a high recovery potential. Many perennial grass seedlings appeared to have established (i.e., seedling growth in June 2006 was the 3rd to 4th leaf stage) using criteria developed by Reis and Svejcar (1991). The reduction and lack of recovery of *P. bulbosa* after fire indicate that this species may be effectively controlled by fire treatments when present in *J. occidentalis* woodlands. Although *B. tectorum* was a minor component of herbaceous response in the Needlegrass association, in areas of litter accumulation (litter mats surrounding stumps and beneath cut trees), where perennial grasses were largely eliminated, *B. tectorum* dominated early succession. Despite presence of *B. tectorum* in these locations, perennial grass seedlings were establishing in these zones in 2006 which was likely aided by above average precipitation in late spring (Sheley and Bates 2008). In other *J. occidentalis* control projects, it has not been unusual for perennial grasses to establish and increase despite enhanced densities and biomass of *B. tectorum* (Vaitkus and Eddleman 1987; Miller and others 2005; Bates and others 2005; Bates and others 2007b; Bates and Svejcar 2009). However, there are cases when *B. tectorum* has dominated following *Pinus-Juniperus* treatment including cutting (Vaitkus and Eddleman 1987), chaining (Tausch and Tueller 1977), fire (Barney and Frischknecht 1974; Bates and others 2007b; Quinsey 1984), slash burning (Haskins and Gehring 2004), and herbicide application (Evans and Young 1985).

Differences in *B. tectorum* response and site recovery after treating *J. occidentalis* and other *Pinus-Juniperus* woodlands is likely linked to species composition and herbaceous perennial density. Bates and others (2005) determined that 2–3 bunchgrasses per m^2 were sufficient to restore the native understory after cutting Phase *3 J. occidentalis* woodlands that were susceptible to *B. tectorum* invasion. In our study, where natives dominated post-fire recovery, densities of perennial grasses were greater than 1–2 plants m^2 and perennial forbs greater than 5 plants m^2 the first year after fire. Dominance by *B. tectorum* is most likely to occur where there is little perennial herbaceous vegetation remaining after juniper treatment (Evans and Young 1985; Bates and others 2007b). Where *B. tectorum* has dominated after fire, post-fire densities of perennial grasses were less than 1 plant m^2 and perennial forbs were less than 5 plants m^2 (Bates and others 2006; Bates and others 2007a, b).

Sagebrush-Steppe Restoration

Partial cutting and prescribed fire combinations are a recent management treatment to remove Phase 3 *J. occidentalis* woodlands and restore shrub-steppe plant communities. Previously, treatments using heavy machinery or chainsaws were the most common practice for managing Phase 3 woodlands because applying controlled fires was problematic (Miller and others 2005). However, mechanical treatment of *J. occidentalis* woodlands is often more expensive compared to prescribed fire treatments. Thus, partial cutting and prescribed fire combinations provide land managers with an additional treatment option for managing other Phase 3 *Pinus-Juniperus* woodlands. One difficulty in applying this treatment is predicting plant community response. Shrub and herbaceous recovery after *J. occidentalis* control is generally predictable when the plant communities are still in the early stages (Phase 1 and 2) of woodland development and contain an intact understory of shrubs and native herbaceous species (Miller and others 2005). In fully developed woodlands (Phase 3), predicting community response is difficult because recovery is often slower and these areas are often susceptible to non-native weed invasion and dominance as a result of depleted native understories. In this study, native annual and perennial vegetation has dominated early succession. In other studies, fall burning partially cut Phase 3 woodlands severely impacted herbaceous understory vegetation, resulting in the nearly complete removal of perennial bunchgrasses and a high percentage of the perennial forbs (Bates and others 2006; Bates and others 2007b). The main understory species to respond in these studies were nonnative biennial weeds, *B. tectorum*, and native annual forbs. Haskins and Gehring (2004) measured a fourfold higher abundance of invasive annuals in burned *Pinus-Juniperus* slash areas compared to unburned slash treatments. In *P. ponderosa* forests of Arizona, invasive species tend to increase in areas of greater fire severity (Bataineh and others 2006; Griffis and others 2001; Sabo and others 2009). To augment herbaceous recovery after fire, particularly on sites with high bunchgrass mortality, seeding should be considered. On Columbiana and Needlegrass associations, seeding was successful at establishing high densities of perennial bunch grasses and forbs (Sheley and Bates 2008).

Herbaceous recovery after *J. occidentalis* control requires patience. Delays of several years are typical before understories fully respond to tree removal (Barney and Frischknecht 1974; Bates and others 2000; Bates and others 2005; Bates and others 2007a; Bates and Svejcar 2009; Koniak 1985). Treated *J. occidentalis* sites with depleted understories have taken six years for perennial grass densities to recover and over 10 years for perennial grass cover to be restored (Bates and others 2005). Bunchgrasses require two to three years to produce large amounts of viable seed following fire or tree cutting (Bates 2005; Bates and others 2009). Recruitment of new bunchgrasses tends to occur the third to fifth year following *J. occidentalis* treatment (Bates and others 2005; Bates and Svejcar 2009). In our study, a similar post-fire response occurred in the Columbiana and Needlegrass associations.

Altering season of prescribed fire application in Phase 3 *J. occidentalis* woodlands has been shown to reduce fire impacts on native plant species. Cutting followed by winter or early spring burning when soils were frozen and/or above field capacity lowered mortality of native plants and resulted in faster understory recovery (Bates and others 2006; Bates and Svejcar 2009). Winter and spring applied fires were not as successful at killing remaining *J. occidentalis* trees and juveniles compared to fall burned treatments because of lower fire severities and higher fuel moisture (Bates and others 2006). Thus, managers should be aware that *Pinus-Juniper* trees will likely reoccupy sites treated by winter-spring burning in a shorter time period than fall burned areas. However, if restoration objectives are to retain some *J. occidentalis* or other *Pinus-Juniper* species on site, then winter and spring burning provides managers with additional treatment options.

Recovery periods of shrub species on both community types will likely take longer than reported in the literature because shrub cover and density were already suppressed by *J. occidentalis* dominance and the fire killed non-sprouting species. Recovery of non-sprouting shrubs, particularly *A. t.* ssp. *vaseyana,* may exceed 40 years due to fire-caused mortality of remaining shrubs and a depleted seed bank. Typical recovery periods for *A. t.* ssp. *vaseyana* canopy cover have been reported to be between 20 and 40 years (Harniss and Murray 1973; Lesica and others 2007; Ziegenhagen and Miller 2009). Sprouting shrubs, especially *S. oreophilus* and *C. nauseous,* will likely recover to pre-invasion conditions faster because there was little fire caused mortality and these species typically increase after fire (Anderson and Bailey 1979; Sieg and Wright 1996; Wright and Bailey 1982). The emergence of *C. velutinus* was probably due to its presence in the seed bank. Viable seed of *C. velutinus* can be stored in soils for at least 200 years (Bradley and others 1992; Kramer and Johnson 1987). Browsing by *Odocoileus hemionus* (mule deer) and *Cervus canadensis nelsoni* (Rocky Mountain elk) was restricting further establishment and growth of *C. velutinous*.

Conclusions

Cutting-prescribed fire combinations are an effective means for controlling encroaching Phase 3 *J.* occidentalis woodlands, and could potentially be applied in other *Pinus*-*Juniperus* woodlands. When burning sites with comparable levels of *J. occidentalis* cover (35–70%) and density (150–475 trees ha^{-1}) a maximum of about one-quarter of the trees should be cut to carry prescribed fire. On similar plant associations, this method of *J. occidentalis* control can initially be expected to stimulate perennial and annual forbs. Perennial grasses will take longer to recover because of reduced densities that are a product of *J. occidentalis* dominance prior to treatment and fire-caused mortality. However, the high mortality sustained by herbaceous perennials, particularly bunchgrasses, increases the potential for cheatgrass or other exotic weeds to dominate areas (Bates and others 2006; Bates and others 2007b). Cutting and burning combinations are not recommended in areas where exotic weeds threaten to dominate after fire. When making site selections for *J. occidentalis* control, managers must carefully consider the potential benefits and problems associated with cutting-prescribed fire applications. If weeds are a threat then winter burning of cut trees or cutting and leaving trees to minimize perennial grass mortality is probably the best option. Because removing Phase 3 woodlands is difficult and expensive and offers less predictable results, managers involved with shrub-steppe restoration should first target *J. occidentalis* control efforts in Phase 1 and Phase 2 successional woodlands. These stages will generally have more intact and complete herbaceous and shrub layers, making restoration efforts more predictable and successful.

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