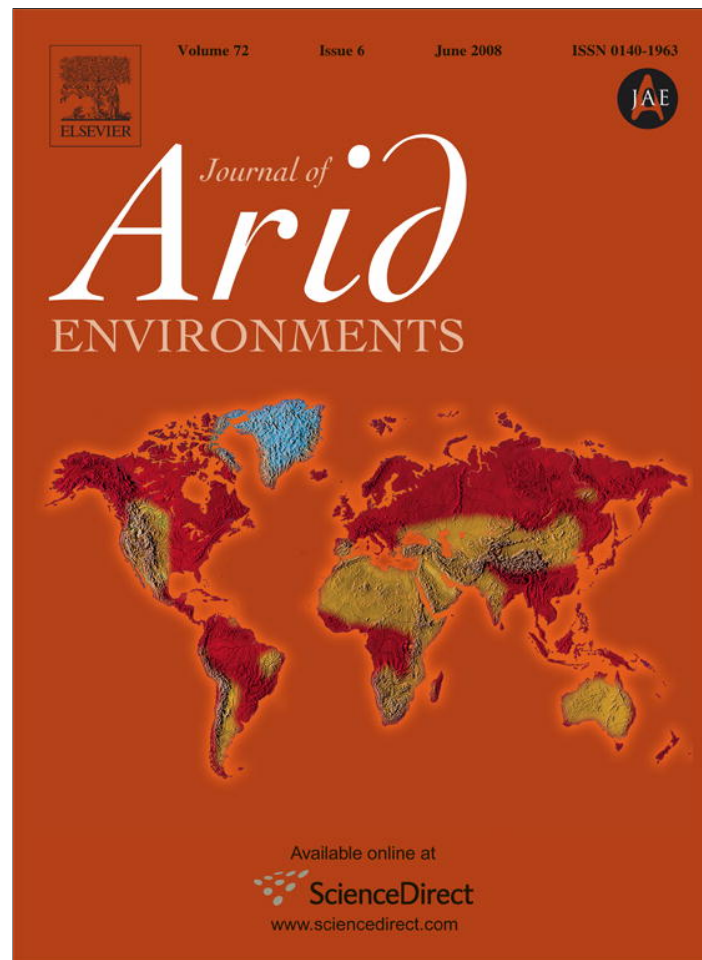


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Does fall prescribed burning *Artemisia tridentata* steppe promote invasion or resistance to invasion after a recovery period? [☆]

Kirk W. Davies^{*}, Roger L. Sheley, Jonathan D. Bates

United States Department of Agriculture (USDA)—Agricultural Research Service at the Eastern Oregon,
Agricultural Research Center, 67826-A Hwy 205, Burns, OR 97720, USA

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Abstract

Artemisia tridentata ssp. *wyomingensis* (Beetle & A. Young) S.L. Welsh-bunchgrass communities were used to analyze the influence of disturbances on invasibility after a recovery period. These communities evolved with periodic fires shifting dominance from shrubs to herbaceous species. However, fire can facilitate *Bromus tectorum* L. invasion of these plant communities. We evaluated the invasibility of *A. tridentata* ssp. *wyomingensis*-bunchgrass communities 4 years after prescribed fall burning at six sites by comparing burned to unburned (control) communities. These communities did not have *B. tectorum* present prior to introduction. *B. tectorum* was introduced at 1, 10, 100, 1000, and 10,000 seeds m⁻² in burned and unburned communities. *B. tectorum* individuals established only when introduced at 10,000 seeds m⁻². In the areas seeded at 10,000 seeds m⁻², *B. tectorum* density and cover were more than three-fold higher in the control than burned treatments ($P = 0.04$ and 0.08 , respectively). Total herbaceous vegetation cover, density, and production increased with burning ($P < 0.01$, 0.02 , and < 0.01 , respectively). Bare ground and inorganic nitrogen were higher in the control than the burned treatment ($P = 0.02$ and < 0.01 , respectively). Prescribed fall burning of late seral *A. tridentata* ssp. *wyomingensis*-bunchgrass communities stimulated the herbaceous component and increased the resistance of the communities to *B. tectorum* invasion 4 years post-burn. However, we do not suggest the use of prescribed burning in communities where invasive annual grasses are present or in close proximity. We acknowledge that our results would probably have been drastically different if *B. tectorum* or other invasive annual grasses had been a component of the plant communities prior to prescribed burning or became a component immediately after burning.

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1. Introduction

The current threat of exotic plant invasions after disturbances requires that ecologists and policy makers understand the influence of prescribed fire-induced disturbances in ecosystems that were historically

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^{*} Corresponding author. Tel.: +1 541 573 4074; fax: +1 541 573 3042.

E-mail address: Kirk.Davies@oregonstate.edu (K.W. Davies).

maintained by periodic disturbances. Historically, *Artemisia tridentata* ssp. *wyomingensis* (Beetle & A. Young) S.L. Welsh (Wyoming big sagebrush)-bunchgrass steppe communities were maintained by periodic wildfires (Wright and Bailey, 1982). Modern day burning, however, may promote the invasion of these communities by *Bromus tectorum* (Stewart and Hull, 1949; Young and Allen, 1997). Understanding how prescribed burning followed by a recovery period (plant community has been allowed to respond to the shift in dominance) affects community susceptibility to *B. tectorum* L. (cheatgrass) invasion will provide information resource managers need to more accurately assess the effects prescribed fire may have in the *A. tridentata* ssp. *wyomingensis* steppe. Also, this information will provide a better understanding of the potential role of disturbances in other ecosystems that historically experienced similar infrequent disturbances.

A. tridentata ssp. *wyomingensis* communities are considered the least resilient and most susceptible of the *A. tridentata* Nutt. (big sagebrush) complex to invasion by exotic weeds (Miller and Eddleman, 2000). Many *A. tridentata* ssp. *wyomingensis*-bunchgrass communities have been converted to *B. tectorum*-dominated annual grasslands, particularly in the Intermountain West due to increased fire frequencies (Miller and Eddleman, 2000; Whisenant, 1990). The majority of the exotic annual grasslands dominated by *B. tectorum* in the Intermountain West were probably formerly *A. tridentata* ssp. *wyomingensis* steppe (Miller and Eddleman, 2000).

The decline of intact *A. tridentata* ssp. *wyomingensis*-bunchgrass communities has generated debate regarding the value and risks associated with using prescribed fire to mimic historic fire regimes. Prescribed burning of *A. tridentata* ssp. *wyomingensis*-bunchgrass steppe has generally been avoided because it is assumed to increase the invasibility of these communities by increasing safe site and resource availability. More safe sites increases the likelihood that propagules of invading species will reach a safe site upon introduction and this could potentially alter succession of the plant community to an undesirable state (Pickett et al., 1987; Sheley and Krueger-Mangold, 2003; Sheley et al., 1996). Of particular concern in *A. tridentata* ssp. *wyomingensis*-bunchgrass steppe is the potential for prescribed burning to promote invasion of exotic annual grasses. Invasion by exotic annual grasses, especially *B. tectorum*, can produce fire return intervals that are too short for reestablishment of *A. tridentata* and are detrimental to desirable herbaceous vegetation (Stewart and Hull, 1949; Whisenant, 1990).

Stewart and Hull (1949) and Young and Allen (1997) have implied that *B. tectorum* rapidly invades *A. tridentata* ssp. *wyomingensis*-bunchgrass communities after fire. However, the majority of *A. tridentata* rangeland they described supported *B. tectorum* and/or were in depleted states prior to the fire. Ecologists and land managers have assumed that burning *A. tridentata*-bunchgrass communities increases resources for *B. tectorum*. The initial impact of the burning usually increases resources (Davies et al., 2007; Hobbs and Schimel, 1984; Young and Allen, 1997), which would favor *B. tectorum* invasion (Young and Allen, 1997). However, Blank et al. (1994) reported decreases in nitrate following wildfire in *A. tridentata* communities, but their study sites were already invaded with *B. tectorum*. Previous studies were conducted immediately post-fire, largely consisting of previously invaded *A. tridentata* communities, and/or did not measure invasibility. Thus, information is lacking describing the influence of prescribed burning followed by a recovery period on the invasibility of late seral *A. tridentata* ssp. *wyomingensis*-bunchgrass communities not already invaded by *B. tectorum*.

Because *A. tridentata* ssp. *wyomingensis*-bunchgrass communities are estimated to have evolved with fire return intervals of 50–100 years (Wright and Bailey, 1982), they may need periodic (~75 years) burns to maintain resistance to invasion. However, current management practices (e.g. wildfire suppression) have or will probably lengthen fire return intervals in some late seral *A. tridentata* ssp. *wyomingensis*-bunchgrass communities. These practices leave the *A. tridentata*-bunchgrass communities in an *A. tridentata*-dominated state, potentially to the detriment of community invasion resistance. Attempts to stabilize a system in one particular vegetation-dominated state by removing natural disturbances often reduces community resilience by eliminating the mechanisms that allow the system to adapt to external change, making them more likely to cross-ecological thresholds and experience dramatic shifts in state (Groffman et al., 2006).

Native plant communities can be seed- or site-limited and this can influence invasibility. For example, Tilman (1997) reported native grasslands in his study were seed-limited. However, Turnbull et al. (2000) in a review of several studies concluded plant communities could be seed- or site-limited. We found no literature detailing whether late seral *A. tridentata* ssp. *wyomingensis*-bunchgrass communities are seed- or site-limited.

However, the invasibility of *A. tridentata* ssp. *wyomingensis*-bunchgrass communities and the implications of longer fire return intervals depend on whether these communities are seed- or site-limited and how burning affects seed and site availability. If microsites are available for occupation, but the invader is not available, then the plant community is seed-limited. However, if the invader is available, but available sites are absent or few then the community is site-limited. Furthermore, the influence prescribed fire has on the long-term site availability is unknown.

Our objective was to determine if re-introducing (prescribing) fire to a late seral, non-invaded *A. tridentata* ssp. *wyomingensis*-bunchgrass community affected invasion potential from *B. tectorum* after the plant community was allowed to recover from fire for three growing seasons. A recovery period following burning was used to allow the desirable herbaceous vegetation to respond to the increase in resources and decrease in competition from sagebrush. A recovery period of 3 years was selected because herbaceous vegetation would probably have already responded to fall prescribed burning influence on resource availability. Davies et al. (2007) had reported a large increase in desirable herbaceous vegetation 2 years after fall prescribed burning similar sites. We hypothesized that after a recovery period (three growing seasons) following prescribed fall burning, that where burning was re-introduced, invasion resistance would be greater than where burning had not occurred. Specifically, we hypothesized that (1) *B. tectorum* would more readily establish in unburned *A. tridentata* ssp. *wyomingensis*-bunchgrass communities than in burned areas where the herbaceous vegetation component had three growing seasons to respond to the burn and (2) resources would be less available in the upper portion of the soil profile of the burned than unburned communities.

2. Methods

2.1. Study area

The study was conducted at the Northern Great Basin Experimental Range (NGBER), in southeast Oregon (43°29'N, 119°43'W) about 56 km west of Burns, OR. The study sites average 300 mm of precipitation annually (Eastern Oregon Agricultural Research Center data file). Precipitation was 117% of the long-term historic record in the 2005–2006 crop year (1 October–30 September) (Eastern Oregon Agricultural Research Center data file). Elevation at the study sites is approximately 1400 m above sea level and topography is relatively flat (slopes < 2°). Soils at the study area are a complex of Haploxerolls, Argixerolls, Durixerolls, and Durargids (Lentz and Simonson, 1986) and surface texture is sandy loam to loamy sand. Sites were not grazed 5 years before the burns and had experienced only light to moderate use (< 40% of the herbaceous biomass removed annually and rarely grazed in 2 consecutive years) for approximately 70 years prior to inclusion in this study. Study sites were late seral *A. tridentata* ssp. *wyomingensis*-bunchgrass communities prior to burning. These plant communities were in high ecological condition without any invasive plants. *A. tridentata* ssp. *wyomingensis* cover ranged between 9% and 15% depending on site prior to burning. Depending on location, either *Achnatherum thurberianum* (Piper) Barkworth (Thurber's needlegrass) or *Pseudoroegneria spicata* (Pursh) A. Löve (bluebunch wheatgrass) is the dominant perennial bunchgrass. *Festuca idahoensis* Elmer (Idaho fescue), *Koeleria macrantha* (Ledeb.) J.A. Schultes (prairie junegrass), *Poa sandbergii* Vasey (Sandberg bluegrass), and *Elymus elymoides* (Raf.) Swezey (squirreltail) are other common perennial bunchgrasses at the study sites. Common forbs include *Crepis* sp. L. (hawksbeard), *Astragalus curvicaulis* (Heller) J.F. Macbr. (curve-pod milkvetch), *Lupinus caudatus* Kellogg (tailcup lupine), *Achillea millefolium* L. (common yarrow), *Phlox longifolia* Nutt. (long-leafed phlox), *Alyssum alyssoides* (L.) L. (desert alyssum), and *Collinsia parviflora* Lindl. (little blue-eyed Mary). *B. tectorum* was not a documented component of these communities prior to our study.

2.2. Experimental design and statistical analysis

A randomized complete block design was used to determine the effects of fall prescribed burning in the fourth post-burn growing season on the resistance of *A. tridentata* ssp. *wyomingensis*-bunchgrass communities to exotic annual grass invasion. Six sites (blocks) were randomly located across an *A. tridentata* ssp. *wyomingensis*-bunchgrass landscape at the NGBER. Blocks varied among one another in soil taxonomy and

dominant herbaceous species, but were relatively homogenous within each block. Each block contained two $50 \times 80 \text{ m}^2$ (0.4 ha) plots randomly assigned to a burned or unburned (control) treatment. This generated six control and six burned treatment plots. Pre-burn vegetation measurements revealed no differences between plots assigned to the burn and control treatments ($P > 0.05$). The burned treatments were applied in October 2002 using a gel-fuel terra torch (Firecon, Inc., Ontario, Oregon). Burns were complete across the burn treatment plots and removed all *A. tridentata* ssp. *wyomingensis* individuals. *A. tridentata* ssp. *wyomingensis* remained absent from the burned treatments throughout the study. During the prescribed burns wind speeds varied between 6 and 20 km h^{-1} , relative humidity varied from 10% to 35%, and air temperatures were 10–25 °C. Moisture of herbaceous vegetation loads averaged 10% and fine fuel loads were between 350 and 420 kg ha^{-1} . Following the prescribed burns, vegetation in the burned treatment had three growing seasons to recover before *B. tectorum* was introduced on 24 February 2006. *B. tectorum* seed source used in the experiment had seed viability of 97% germination and was collected from a site adjacent to the east boundary of the NGBER. *B. tectorum* was broadcast seeded on the soil surface in five 1-m^2 plots in each treatment in each block. The 1-m^2 plots were randomly selected to be seeded at 1, 10, 100, 1000, or 10,000 seeds m^{-2} . Introducing *B. tectorum* at different seed rates was used to determine the availability of sites and ascertain what level of introduction poses a potential threat to the existing plant community. Because *B. tectorum* only successfully established (at least one individual successful completed its life-cycle) when seeded at the 10,000 seeds m^{-2} rate, only that seeding rate was analyzed between the burned and control treatments to compare invasibility. However, in a separate study on the same six study sites in 2006, *B. tectorum* introduced at 100 and 1000 seeds m^{-2} established when perennial vegetation was removed. *B. tectorum* density was 50 ± 3.4 (mean \pm SE) and 523 ± 25.6 individuals m^{-2} when introduced at 100 and 1000 seeds m^{-2} , respectively. Perennial vegetation had been removed by cutting the plants below the crown with a soil knife or saw and removing the above ground biomass from the treatment plots in October 2005. Perennial plants that were not successfully removed in October were removed when detected in 2006. Response variables were *B. tectorum* density and cover, other herbaceous cover, density, and biomass, bare ground cover, and soil water, phosphorus, potassium, and inorganic nitrogen in the upper 20 cm of the soil profile.

Randomized complete block analyses of variance (ANOVA) were used to test for treatment differences between response variables that were not repeatedly sampled across the growing season (SAS Institute, 2001). Differences between means were considered significant if P -values were less than 0.10 ($\alpha = 0.10$). Repeated-measures ANOVA was used for variables that involved repeated sampling through the growing season (SAS Institute, 2001). Between-subject effects were block and treatment. Within-subject effects were sampling date and the interactions of sampling date with the between-subject effects. Data that did not meet ANOVA assumptions of normality were log-transformed.

2.3. Measurements

B. tectorum density and cover were measured in late June. *B. tectorum* density was measured in each 1-m^2 plot. *B. tectorum* foliar cover was visually estimated in each 1-m^2 plot. Herbaceous vegetation cover, density, and biomass production were measured across each treatment plot. Herbaceous biomass was determined in late June by clipping, oven drying, and then weighing the current year's growth from 15 randomly located 1-m^2 frames per treatment plot. Herbaceous cover and density were measured along five, 50-m transects spaced at 20 m intervals deployed along an 80 m main transect in each treatment plot. Herbaceous canopy cover and density were recorded by species inside 0.2 m^2 frames ($40 \times 50 \text{ cm}^2$) located at 3 m intervals on each transect line (starting at 3 m and ending at 45 m), resulting in 15 frames per transect and 75 frames per plot. Herbaceous canopy cover was visually estimated in each 0.2 m^2 frame.

Soil water content was measured gravimetrically every 2 weeks from early April until the end of June. Five 2 cm diameter soil cores (0–20 cm) were collected in each treatment plot. Each soil core was weighed, dried at 100 °C to a constant weight, and then weighed again to determine soil water content. To estimate nutrient supply rates of potassium, phosphorus, and inorganic nitrogen (nitrate and ammonium) between treatments, four anion and cation PRSTM-probes (Western Ag Innovations, Saskatoon, Saskatchewan, Canada) were randomly placed in each treatment plot. These PRSTM-probe pairs were buried directly into the soil to estimate the availability of soil nutrients to plants in each treatment plot (Jowkin and Schoenau, 1998).

PRSTM-probes attract and adsorb ions through electrostatic attraction on an ion-exchange membrane. The PRSTM-probes were placed vertically in the upper 20 cm of the soil profile to estimate nutrient supply rates. The PRSTM-probes were buried from the May 1st until June 15th. PRSTM-probes were returned to Western Ag Innovations for analysis. The probes were extracted with 0.5 HCl and analyzed colourimetrically with an autoanalyzer.

3. Results

3.1. Vegetation

B. tectorum did not establish when seeded at 1000 seeds m⁻² rate or less. At the 10,000 seeds m⁻² rate, *B. tectorum* establishment was greater in the control than the burned treatment. *B. tectorum* density was more than three-fold greater in the control than burned treatments ($P = 0.04$) (Fig 1). *B. tectorum* foliar cover was also more than three-fold greater in the control than burned treatments ($P = 0.08$).

Herbaceous vegetation cover, density, and biomass were greater in the burned treatment compared with the control treatment. Total herbaceous cover was 1.5-fold greater in the burned than control treatments ($P < 0.01$) (Fig. 2). Total perennial herbaceous cover was 1.4-fold greater in the burn compared with the control ($P = 0.01$). Annual herbaceous cover was also greater in the burn than control treatment ($P = 0.08$). Bare ground was greater in the control than the burned treatments ($P = 0.02$). Total and annual herbaceous density were about three-fold higher in the burned than control treatments ($P = 0.02$ and 0.02 , respectively) (Fig. 3A). Total perennial herbaceous density did not differ between treatments ($P = 0.55$). Total and perennial herbaceous production values were 1.8- and 1.7-fold greater in the burned than control treatments ($P < 0.01$) (Fig. 3B). Mean annual herbaceous production was 1.9-fold greater in the burned than control treatment ($P = 0.07$).

3.2. Soil

Soil water content in the upper 20 cm did not differ between treatments ($P = 0.77$), but varied over time ($P < 0.01$) (Fig. 4). Over the growing season soil water generally decreased. Treatment effects did not vary with time ($P = 0.22$). Inorganic soil nitrogen was 1.6-fold greater in the control treatment compared with the

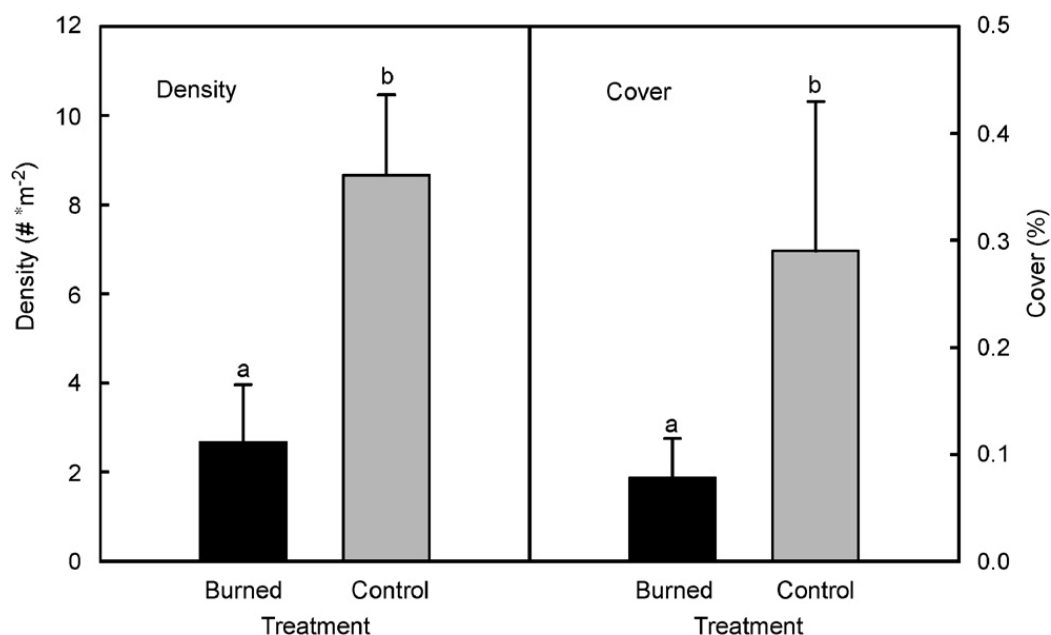


Fig. 1. *B. tectorum* density and cover (mean + SE) in the burned and control treatments in *A. tridentata* ssp. *wyomingensis*-bunchgrass communities 4 years post-burn. Lower case letters indicate significant differences between treatments ($P < 0.10$).

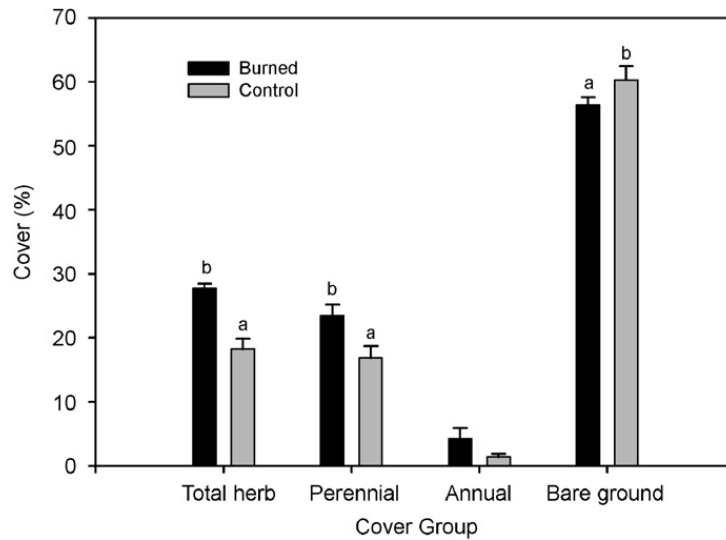


Fig. 2. Herbaceous and bare ground cover (mean+SE) for burned and control treatments in *A. tridentata* ssp. *wyomingensis*-bunchgrass communities 4 years post-burn. Total herb = total herbaceous, perennial = perennial herbaceous, and annual = annual herbaceous. Lower case letters indicate significant differences between treatments ($P < 0.10$).

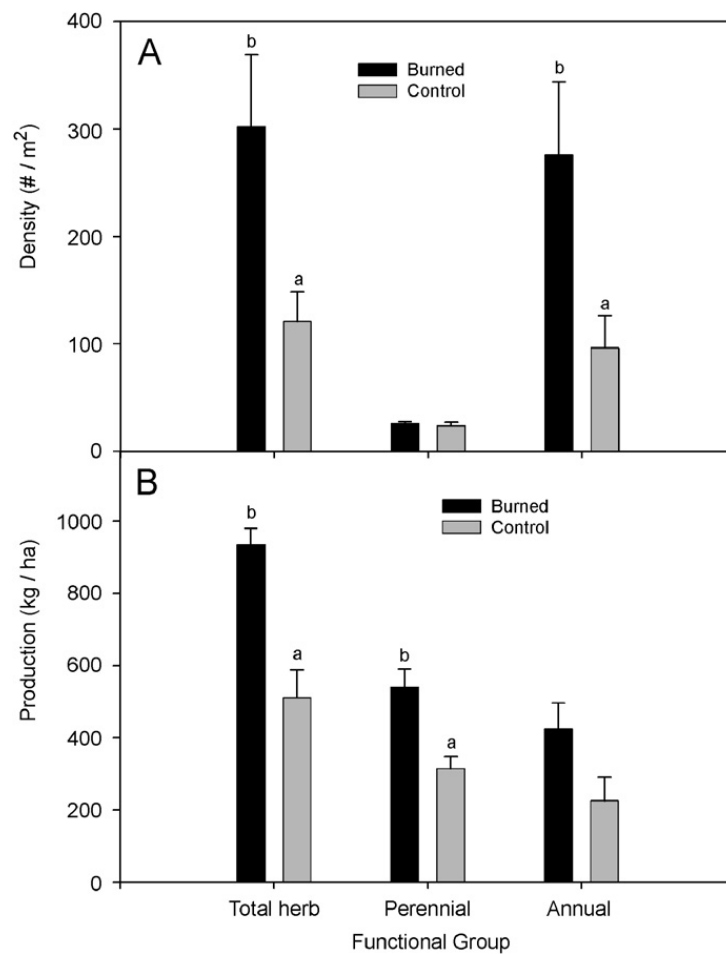


Fig. 3. (A) Herbaceous density (mean+SE) in the burned and control treatments in *A. tridentata* ssp. *wyomingensis*-bunchgrass communities 4 years post-burn. (B) Herbaceous production (mean+SE) for burned and control treatments in *A. tridentata* ssp. *wyomingensis*-bunchgrass communities 4 years post-burn. Total herb = total herbaceous, perennial = perennial herbaceous, and annual = annual herbaceous. Lower case letters indicate significant differences between treatments ($P < 0.10$).

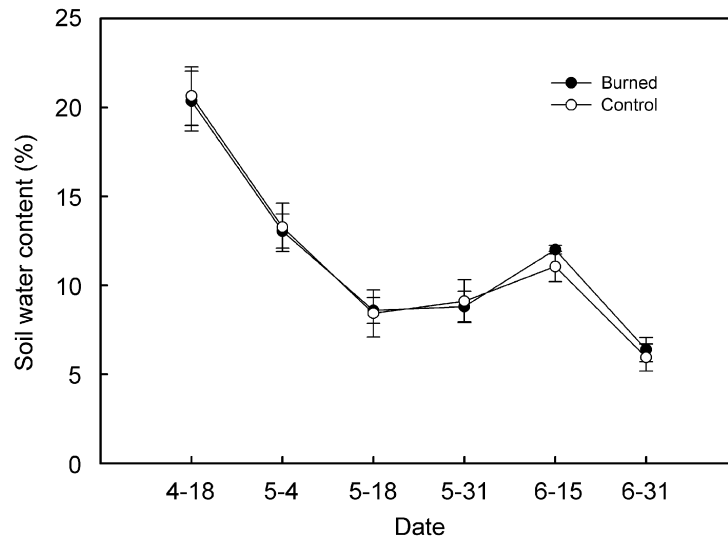


Fig. 4. Soil water content (mean \pm SE) in the upper 20 cm of the soil profile for the control and burned treatments in *A. tridentata* ssp. *wyomingensis*-bunchgrass communities 4 years post-burn. Lower case letters indicate significant differences between treatments ($P < 0.10$).

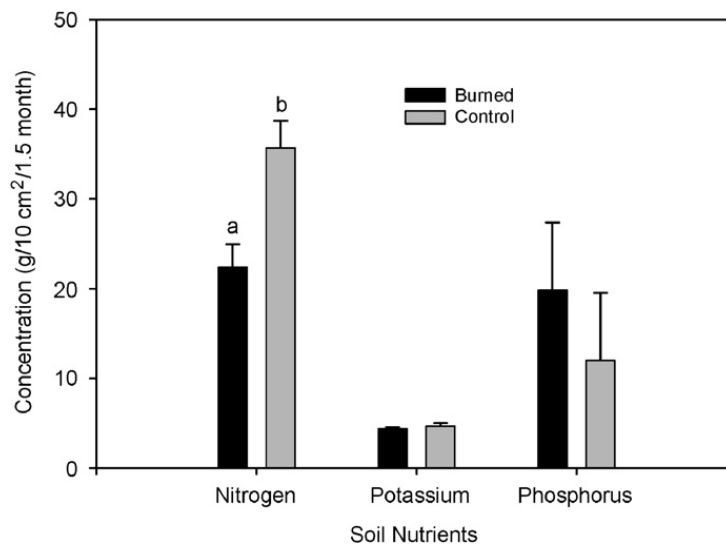


Fig. 5. Soil potassium, phosphorus, and inorganic nitrogen concentrations (mean \pm SE) in the upper 20 cm of the soil profile over the growing season for the burned and control treatments in *A. tridentata* ssp. *wyomingensis*-bunchgrass communities 4 years post-burn. Lower case letters indicate significant differences between treatments ($P < 0.10$).

burned treatment ($P < 0.01$) (Fig. 5). Soil phosphorus and potassium did not differ between treatments ($P = 0.36$ and 0.36 , respectively).

4. Discussion

Our results suggest that periodic disturbances may be needed to maintain the long-term invasion resistance of plant communities that evolved with disturbances. In our study, prescribed fall burning increased the resistance of *A. tridentata* ssp. *wyomingensis*-bunchgrass steppe to *B. tectorum* invasion when the plant community had a 3-year recovery period prior to *B. tectorum* seed introduction. The greater *B. tectorum* density and cover, as well as, the higher concentrations of soil inorganic N in the control compared with the burned treatment suggests burning increased invasion resistance in the burned sites. *B. tectorum* invasion is closely linked with increased nitrogen (Beckstead and Augspurger, 2004; Young and Allen, 1997) and, thus by decreasing soil inorganic N concentrations with burning followed by a short recovery period, invasibility was

also potentially decreased. Kolb et al. (2002) and Seabloom et al. (2003) also reported increased N promoted exotic plant invasion.

The greater herbaceous vegetation cover, density, and production in the burned than control treatment may have interfered with the ability of *B. tectorum* to invade the burned treatment. Increasing desirable vegetation can decrease the susceptibility of a plant community to weed invasion (Knight and Reich, 2005; Smith and Knapp, 1999; Troumbis et al., 2002). The burned treatment also exhibited less bare ground than the control treatment. Burke and Grime (1996) reported that the invasibility of plant communities were positively correlated to the amount of bare ground. The increase in desirable vegetation and decrease in bare ground probably reduced site and resource availability and thus decreased *B. tectorum* establishment.

Because *A. tridentata* ssp. *wyomingensis*-bunchgrass communities are estimated to have evolved with fire return intervals of 50–100 years (Wright and Bailey, 1982), they may need periodic prescribed burns to maintain herbaceous composition and resistance to invasion. Stabilizing *A. tridentata*-bunchgrass communities in an *A. tridentata*-dominated state may eliminate or decrease herbaceous components critical to the system. Removing natural disturbances to sustain an ecosystem in one particular state often reduces its ability to adapt to external change and increases the likelihood of irreversible changes (Groffman et al., 2006). For example, suppressing fires in *A. tridentata* ssp. *vaseyana* (Rydb.) Beetle (mountain big sagebrush)-bunchgrass communities has resulted in encroachment of *Juniperus occidentalis* Hook (western juniper) (Miller and Rose, 1995, 1999). Periodic fires are critical to maintain *A. tridentata* ssp. *vaseyana* communities and our results imply that they may also be important to *A. tridentata* ssp. *wyomingensis* communities. This suggests other plant communities that evolved with disturbances may need those disturbances, at least infrequently, to sustain themselves and to protect them from irreversible changes.

Our results combined with previous studies (Stewart and Hull, 1949; Young and Allen, 1997) imply that the results of prescribed burning *A. tridentata*-bunchgrass communities to *B. tectorum* invasion may vary with the status of the post-fire vegetation at the time of introduction of *B. tectorum* seed. Fire often temporarily increases resource availability (Davies et al., 2007; Hobbs and Schimel, 1984; Young and Allen, 1997) and resistance to *B. tectorum* invasion decreases with increased resource availability (Beckstead and Augspurger, 2004). Stewart and Hull (1949) reported that *B. tectorum* rapidly invades recently burned sites; however, most of their observations involved depleted states and/or had *B. tectorum* or its seeds available prior to the fire. *B. tectorum* was not present at our study sites and the native herbaceous understory vegetation was intact. Furthermore, our fall prescribed burns resulted in limited mortality of herbaceous vegetation and the herbaceous plant and soil communities were allowed to respond to the release of resources prior to *B. tectorum* introduction. Results may have been different if the herbaceous communities were exhibiting negative impacts from prescribe burning when *B. tectorum* seeds were introduced or if *B. tectorum* was already present at the sites prior to burning.

The absence of *B. tectorum* in the burned and control treatments seeded at 1, 10, 100, and 1000 seeds m⁻² suggests very few microsites are available for *B. tectorum* to occupy in late seral *A. tridentata* ssp. *wyomingensis*-bunchgrass steppe. Further supporting this assumption is the high rate of establishment of *B. tectorum* when all perennial vegetation was removed. These plant communities appear to be site-limited rather than seed-limited. After a short recovery period, prescribed fall burning appears to reduce safe site availability. These results imply that most introductions of *B. tectorum* into late seral *A. tridentata* ssp. *wyomingensis*-bunchgrass steppe will fail because the likelihood of *B. tectorum* seeds dispersing to an available safe site is small. Our results are contrary to what Tilman (1997) reported for a native grassland, but Turnbull et al. (2000) in their review of the literature reported many plant communities had no evidence of seed limitation.

5. Conclusions

Disturbances are important in maintaining ecosystems that evolved with disturbances and after a recovery period may increase the resistance of plant communities to exotic plant invasions. Infrequent prescribed burning can improve the resistance of *A. tridentata* ssp. *wyomingensis*-bunchgrass communities to *B. tectorum* and potentially other exotic plant invasions. However, the plant communities burned in this study were not invaded by *B. tectorum* prior to burning, had a relatively intact native herbaceous understory, and suffered limited mortality of perennial herbaceous vegetation from the prescribed fall burn. Our results should not be

extrapolated to already depleted plant communities, locations infested with invasive plants, or landscapes that did not evolve with large-scale disturbances. The initial increase in resource availability from prescribed burning *A. tridentata* ssp. *wyomingensis*-bunchgrass communities (Davies et al., 2007) may decrease the resistance of the plant community in the short-term (Beckstead and Augspurger, 2004); however, if invasive plants are not available to establish at this time, prescribed fires can increase the ability of the community to repel future invasion attempts as resource and safe site availabilities are decreased. Other plant communities may exhibit a similar pattern of resource and safe site availabilities following re-introduction of disturbances, thus re-introducing disturbances may also increase their resistance to invasion after a recovery period. Caution should be exercised when planning to re-introduce disturbances into plant community because of the threat of exotic plant invasion. Exotic plant species presence or their ability to disperse to the plant community must be carefully analyzed prior to implementing a prescribed disturbance. Ensuring exotic invasive plants do not disperse to the site before desirable vegetation has reduced the initial flush of resources from the disturbance is needed to reduce the likelihood of invasive plants becoming established. The long-term influence of prescribed disturbances on invasion resistance of other plant communities that are experiencing extended disturbance return intervals should be investigated. This information would increase the ability of ecologists and land managers to improve the resistance of native plant communities to exotic plant invasions.

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