Effect of prescribed fire on a shrub-steppe plant community infested with *Bromus tectorum*.

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Abstract:

Prescribed fire has been used to manipulate plant communities and is used to remove plants and litter for efficient application of pre-emergent herbicides to the soil. An experiment was conducted to determine if prescribed fire in the fall would have a significant impact on plant community structure in the shrub-steppe. Fire effects were compared with those in unburned plots. Fire increased frequency of *Descurainia* species only in the year after the fire. Cover of *Bromus tectorum* was reduced, while cover of *Sisymbrium altissimum* and *Poa secunda* increased in the year after the fire. These effects were gone in the second year. Soil cover increased and litter cover decreased in both years after the fire. There was no effect of the fire on native species cover or richness. The index of very abundant species and the evenness index were greater in burned plots than in unburned plots only in the year after the fire. Prescribed fire, under the conditions of the test, does not have a lasting effect on native flora.

Introduction:

Bromus tectorum, an invasive alien annual grass, has become dominant in many areas of the western United States (Mack 1981). Because *B. tectorum* forms a continuous fuel load and increases the length of the fire season (Pellant 2002), its presence leads to both increased fire frequency (D'Antonio and Vitousek 1992; Stewart and Hull 1949; Whisenant 1990) and to larger fires (Knapp 1998). These fires lead to a reduction in the diversity and cover of native species (Monsen 1994) which allows *B. tectorum* to increase its presence (Knick and Rotenberry 1997). This unfortunate feedback loop has only become stronger in the Intermountain West with many unpleasant consequences for society.

Wildfires have significant effects on shrub-steppe communities especially when *B. tectorum* is common. Areas dominated by *B. tectorum* with high fire frequency show reductions in species diversity (Billings 1990). Daubenmire (1975) found species richness reduced after a fire in a shrub-steppe community having about 95% *B. tectorum* cover. When there was only 11.8% *B. tectorum* cover, Akinsoji (1988) found shrub cover was significantly reduced after a fire, but cover of other fire resistant species of forbs and grasses increased compared with an adjacent unburned area. Low *B. tectorum* cover likely reduces maximum temperatures during a fire (Bond and Wilgen 1996; Whelan 1995), thus reducing negative effects on many species

(Bailey and Anderson 1978; Bond and Wilgen 1996). We examined the effect of a prescribed fire on species composition and cover when *B. tectorum* cover was about 50%.

Prescribed fires are used as an element in restoration (Whelan 1995) in the shrub-steppe. The amount of pre-emergent herbicide can be reduced if plant matter can be removed using a prescribed fire (Link et al. 2004). Prescribed fires are also used to reduce fuel loads to reduce fire risk (Whelan 1995), although frequent prescribed fire can increase the risk of plant species extinction (Bradstock et al. 1998). We test the hypothesis that a prescribed fire applied in the fall will not have an effect on species composition and cover when *B. tectorum* cover is near 50%.

The purpose of this work was to determine if prescribed fire has any strong and negative effects on the native plant community and strong effects on the invasive plant community. This information will be helpful in designing restoration strategies. We conducted an experiment along the edge of a prescribed burn with unburned plots within 3 m of burned plots. Thus, plots are pseudoreplicates (Hurlbert 1984). Many fire studies are limited by such pseudoreplication (Akinsoji 1988; Antos et al. 1983; Daubenmire 1975). Implications of this study are limited to the local area, but are compared with similar studies.

Methods:

Study area

The study area is on the Columbia National Wildlife Refuge (CNWR) in Grant County, Washington. The CNWR is semi-arid with most precipitation (annual average 203 mm) in the fall and winter. Snowfall is variable with winter high temperatures usually near freezing. Lightning during the summer is a frequent cause of wildfire. Slope of the study area were about 10°. Aspect is about 270°.

Plant communities at the CNWR have been dominated by *Artemisia tridentata*, *Pseudoroegneria spicata*, and *P. secunda* and are classified as an *A. tridentata – P. spicata* association (Daubenmire 1970). Cattle and sheep were introduced in the 1800's. The area was severely overgrazed and soon dominated by *B. tectorum*. Grazing was halted more than 20 years ago and fire is the most prominent disturbance to upland areas. The study area burned in 1993. *Bromus tectorum* remains the dominant cover, with variable amounts of annual and perennial forbs. All areas have *Poa secunda* (Sandberg's bluegrass).

Experimental design and Treatment

Six prescribed burn and six unburned control plots were established along the edge of the burn. Dimensions of each plot were about 16.5 m wide and about 32.9 m long.

The plots were burned in the early afternoon on October 1, 2002. The prescribed burn was conducted as a flanking fire, with some backing, and brief periods of head fire.

Measurements

Measurements were taken before the fire in 2002 and repeated in 2003 and 2004. The response variables are species richness and cover. Species richness was determined in each plot by identifying all vascular plant species. This was done by inspection. Inspections occurred in two periods (late March and early April, late May and June) in each year of the study.

Percent cover of each vascular plant species, bare soil, soil cryptogams, and litter was determined in May and June in each year of the study. Cover was determined using a tape (Bonham 1989; Elmore et al. 2003; Link et al. 2005) and identifying the first observed (tallest) cover type at each 0.5 m hash mark on the tape. The tape was stretched tightly between two pieces of rebar metal stakes in the ground at the ends of any transect. Three tape transects were observed along the long dimension in each plot, resulting in at least 191 observations.

Data analysis

Species frequency is the proportion of plots with a particular species present. Frequency of each species was determined separately for the control and burned plots. Two diversity and one evenness index (Stohlgren et al. 1999) were used to assess response. As described in Ludwig and Reynolds (1988) and Stohlgren et al. (1999), N1 is an index of the number of abundant species. N2 is an index of the number of very abundant species. As in Stohlgren et al. (1999), we determined cover as a measure of abundance, thus N1 and N2 refer to high and very high cover species. High N1 and N2 values indicate higher dominant species diversity.

N1 is:

$$\mathrm{N1} = e^{H'}, \tag{1}$$

where Shannon's index (H') is:

$$\mathrm{H'} = -\sum_{i=1}^{S}\left(\left(\frac{n_i}{n}\right)\ln\left(\frac{n_i}{n}\right)\right), \tag{2}$$

and where n_i is cover of the ith species of S species and n is total cover of all species in a plot.

N2 is:

$$\mathrm{N2} = 1/\lambda, \tag{3}$$

where λ (Simpson's index) for a plot is :

$$\lambda = \sum_{i=1}^{S}\frac{n_i(n_i-1)}{n(n-1)}. \tag{4}$$

The E5 evenness index (Ludwig and Reynolds 1988; Stohlgren et al. 1999) is:

$$\mathrm{E5} = \frac{N2-1}{N1-1}. \tag{5}$$

Low evenness indicates one species is increasingly dominant. High evenness indicates greater equivalency in cover among species.

Statistical analysis

Analyses were done using JMP version 5, software (SAS Institute, 2002). Percentage data, when below 20%, were transformed using arcsin $\sqrt{p}$, where p is a proportion, before analysis (Steele and Torrie 1960). Means and error terms are presented using percent data for interpretation. Because response variables were measured on the same plots across years, we used repeated measure analysis of variance (MANOVA). We test for between-subject (fire treatment), within-subject (time), and fire*time interaction effects. Student's t-test was used to compare treatments within a year. Error terms are one standard error of the mean. Statistical significance is at $\alpha = 0.05$.

Results

There were 40 species observed in the plots; eight are alien, 32 are native, 20 are annuals or biennials, and 20 are perennials (Table 1). Species with large changes in frequency across years include *L. serriola* that was not observed until 2004 when it became present in 10 of the 12 plots (Table 1). *Stephanomeria paniculata* was not observed until 2004 when it appeared in 7 of the plots. *Microseris troximoides*, *T. dubius*, *V. microstachys* increased in frequency over the three years and were not influenced by the fire. The two *Descurainia* species were more frequent in 2003 than in 2002 or 2004. They were found in nearly all the fire plots and present in only 33% of the control plots in 2003. *Astragalus* cf. *lentiginosus* and *E. brachycarpum* became more frequent in 2004 than earlier and were not influenced by the fire.

Table 1. Plant species found in study plots at the Columbia National Wildlife Refuge. Note symbols are "n" for native and "a" for alien; "a" for annual, "ab" for annual/biennial, "p" for perennial; "h" for herbaceous, "s' for shrub, and "ss" for sub-shrub. Specific epithets follow that in Hitchcock and Cronquist (1976) and current convention (http://plants.usda.gov/).

Family *Species*	**Frequency Control (n = 6)**			**Frequency Fire (n = 6)**			**Notes**
	2002	**2003**	**2004**	**2002**	**2003**	**2004**	
Boraginaceae							
Amsinckia tessellata	1	1	1	1	1	1	n, a, h
Lappula redowskii	0	0.17	0	0	0.17	0	n, a, h
Caryophyllaceae							
Holosteum umbellatum	1	1	1	1	1	1	a, a, h
Chenopodiaceae							
Salsola tragus	0	0	0.17	0	0	0	a, a, h
Compositae							
Achillea millifolium	0.5	0.67	0.5	0.5	0.33	0.67	n, p, h
Agoseris sp.	0	0	0.17	0.17	0	0	n, p, h
Agoseris grandiflora	0	0	0.17	0	0.17	0.17	n, p, h

Family *Species*	**Frequency Control (n = 6)**			**Frequency Fire (n = 6)**			**Notes**
	2002	**2003**	**2004**	**2002**	**2003**	**2004**	
Agoseris heterophylla	0	0	0.5	0.17	0.17	0.17	n, a, h
Chondrilla juncea	0.17	0.17	0.17	0	0.17	0.17	a, p, h
Conyza canadensis	0.17	0	0	0	0	0	n, a, h
Crepis atribarba	0.5	0.33	0.33	0.5	0.5	0.5	n, p, h
Ericameria nauseosa	0.5	0.5	0.5	0.17	0	0	n, p, s
Erigeron pumulus	0	0	0	0	0	0.17	n, p, h
Lactuca serriola	0	0	1	0	0	0.67	a, a, h
Lagophylla ramosissima	0	0	0.33	0	0	0.17	n, a, h
Machaeranthera canescens	0	0	0	0.17	0	0.17	n, b, h
Microseris troximoides	0	0.33	0.83	0.17	0.33	1	n, p, h
Stephanomeria paniculata	0	0	0.5	0	0	0.17	n, a, h
Taraxacum officinale	0.67	0.33	0.33	0.33	0.5	0.33	a, p, h
Tragopogon dubius	0.83	1	1	0.5	1	1	a, a, h
Cruciferae							
Descurainia pinnata	0.17	0.33	0	0	1	0.17	n, ab, h
Descurainia richardsonii	0.17	0.33	0	0.17	0.83	0.17	n, ab, h
Sisymbrium altissimum	1	1	1	1	0.83	1	a, a, h
Graminae							
Bromus tectorum	1	1	1	1	1	1	a, a
Hesperostipa comata	0.67	0.67	0.67	0.67	0.67	0.67	n, p
Poa bulbosa	0.17	0.17	0.17	0.17	0.17	0.17	a, p
Poa secunda	1	1	1	1	1	1	n, p
Pseudoroegneria spicata	0.83	1	1	0.83	0.83	0.83	n, p
Vulpia microstachys	0.33	0.83	0.83	0.33	0.5	1	n, a
Vulpia myuros	0	0	0.17	0	0	0.17	n, a
Leguminosae							
Astragalus cf. *lentiginosus*	0	0.33	0.5	0.33	0.17	0.67	n, p, h
Liliaceae							
Calochortus macrocarpus	0	1	1	0.17	1	0.83	n, p, h
Onagraceae							
Epilobium brachycarpum	0	0	1	0.33	0	1	n, a, h
Plantaginaceae							
Plantago patagonica	0	0	0	0.17	0	0	n, a, h
Polemoniaceae							
Phlox longifolia	1	1	1	1	1	1	n, p, ss
Portulaceae							
Montia perfoliata	0.17	0.33	0.33	0	0.17	0	n, a, h
Ranunculaceae							
Delphinium nuttallianum	1	1	1	1	1	1	n, p, h
Umbelliferae							

Family *Species*	Frequency Control (n = 6)			Frequency Fire (n = 6)			Notes
	2002	**2003**	**2004**	**2002**	**2003**	**2004**	
Lomatium macrocarpum	1	1	1	1	1	1	n, p, h
Lomatium triternatum	0.83	0.83	0.83	1	1	1	n, p, h
Valerianaceae							
Plectritis macrocera	0	0.33	0.17	0	0	0	n, a, h

Cover and diversity indices

Bromus tectorum cover was significantly (p = 0.0044) less in burned plots than in unburned controls in the growing season (2003) after the fire. The effect of the fire was gone in 2004 (Fig 1a). There is a significant effect of time (p = 0.0001) and interaction between observation time and the fire treatment (p = 0.0012). *Bromus tectorum* cover decreased from about 50% in unburned plots in 2002 and 2003 to about 24% in 2004 (Fig. 1a).

Sisymbrium altissimum cover was significantly (p = 0.0006) greater in burned plots than in unburned controls in the growing season (2003) after the fire. The effect of the fire was gone in 2004 (Fig 1b). There is no effect of time (p = 0.76) or significant (p = 0.57) interaction between observation time and the fire treatment (Fig 1b).

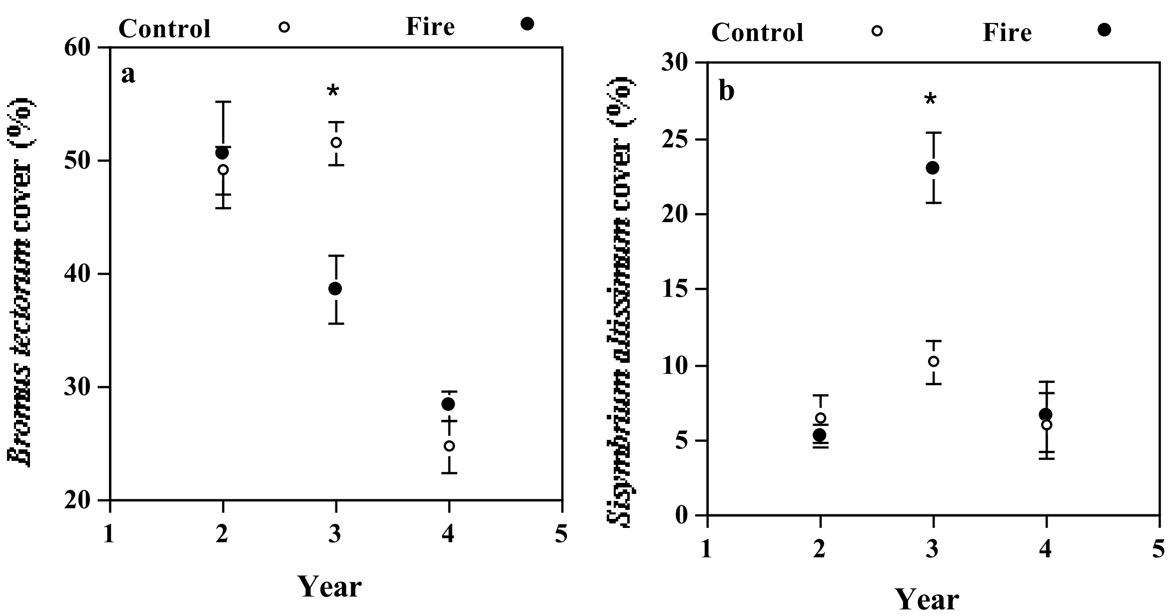

Figure 1. Percent cover of *Bromus tectorum* (a) and *Sisymbrium altissimum* (b) starting in 2002. Bars are one standard error of the mean (n = 6). An asterisk indicates the means in that year are significantly different.

Poa secunda cover was significantly (p = 0.0346) greater in burned plots than in unburned controls in the growing season (2003) after the fire. The effect of the fire was gone in 2004 (Fig 2a). There is a significant effect of time (p = 0.0002) and interaction between observation time and the fire treatment (p = 0.0133).

There was no effect of fire on *A. tessellata* cover (p = 0.415). Cover increased significantly (p < 0.0001) in time (Fig. 2b). There was no significant (p = 0.2161) interaction between observation time and the fire treatment (Fig. 2b).

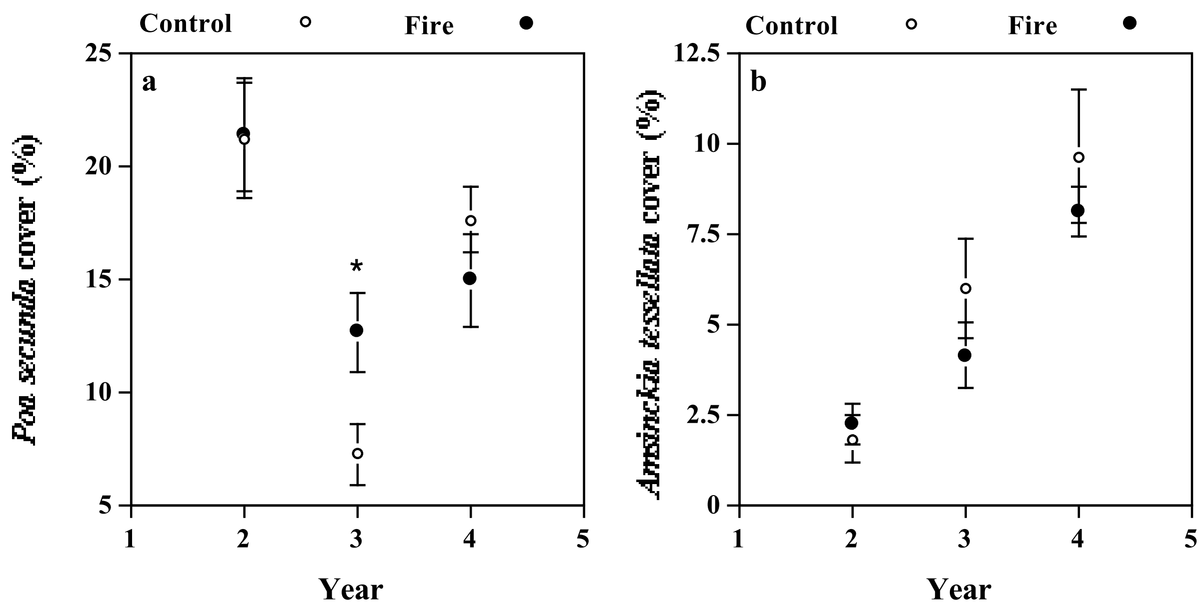

Figure 2. Percent cover of *Poa secunda* (a) and *Amsinckia tessellata* (b) starting in 2002. Bars are one standard error of the mean (n = 6). An asterisk indicates the means in that year are significantly different.

Cover of *E. brachycarpum* was zero in 2002, 0.1% in 2003 in control plots, but increased significantly (p = 0.0002) to 9.4 ± 2.0 % in control plots and to 5.4 ± 1.0% in burned plots in 2004. There was no significant (p = 0.157) effect of fire on *E. brachycarpum* cover in 2004. *Vulpia microstachys* cover significantly (p = 0.0104) increased, linearly, from 0.2 ± 0.1 % in both sets of plots in 2002 to 1.3 ± 0.5 % in control and 0.9 ± 0.2 in burned plots in 2004. There was no significant (p = 0.88) effect of fire on *V. microstachys* cover in 2004.

Soil cover (Fig. 3a) was significantly (p = 0.0101) greater in burned plots than in unburned controls in 2003 after the fire and in 2004 (p = 0.0048). There was a significant effect of time (p < 0.0001) and interaction between observation time and the fire treatment (p = 0.0198).

Litter cover (Fig 3b) was significantly (p = 0.0067) less in burned plots than in unburned controls in 2003 after the fire and in 2004 (p = <0.0001). There was a significant effect of time (p < 0.0071) and interaction between observation time and the fire treatment (p = 0.0006).

Alien and native species cover was not significantly affected by the fire in 2003 or 2004 (Figs. 4a,b). For alien species, there is a significant effect of time (p < 0.0001) and no significant (p = 0.40) interaction between time and fire treatments. For native species, there is a significant effect of time ($p < 0.0001$) and a significant ($p = 0.0184$) interaction between observation time and the fire treatment. In unburned control plots, alien cover dropped from about 66% in 2003 to about 32% in 2004. Native species cover increased from 2003 to 2004 (Fig, 4b).

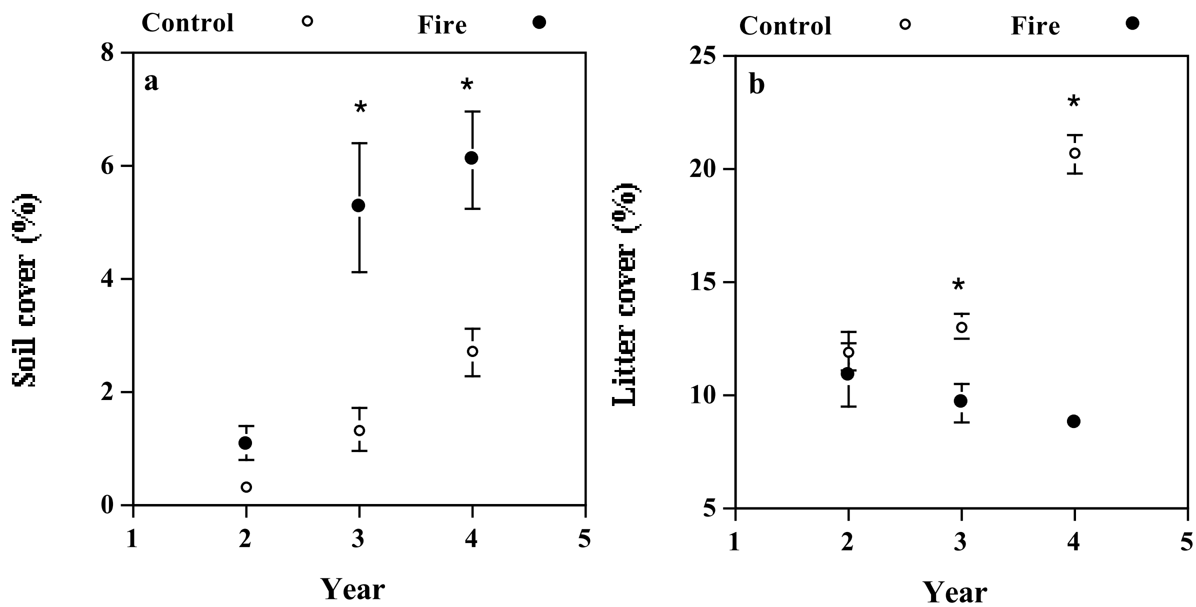

Figure 3. Percent cover of soil (a) and litter (b) starting in 2002. Bars are one standard error of the mean (n = 6). An asterisk indicates the means in that year are significantly different.

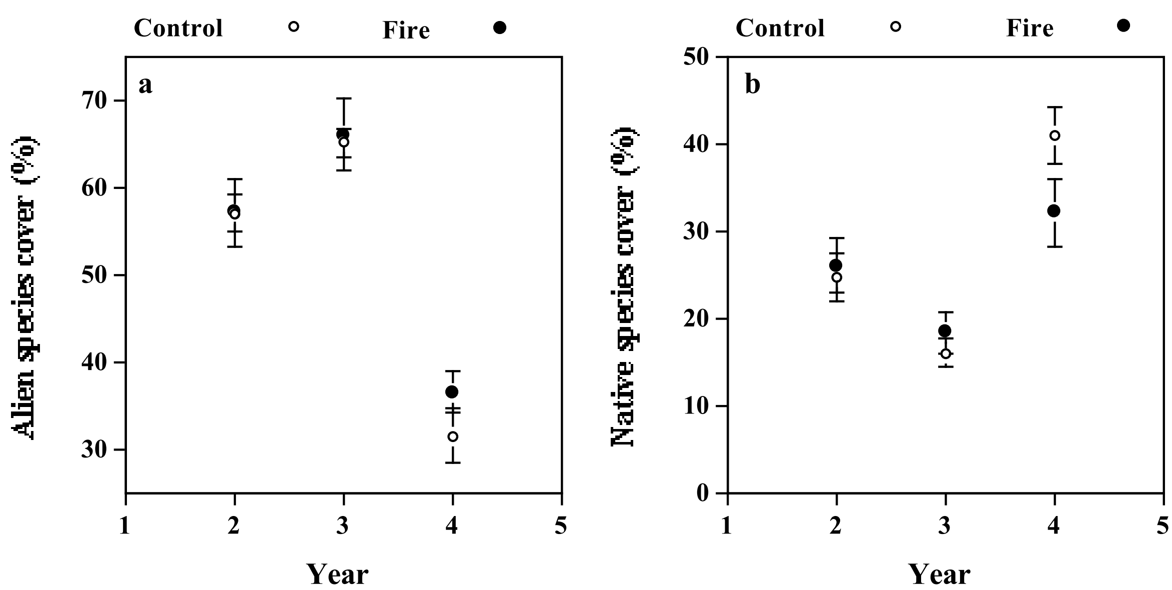

Figure 4. Percent cover of alien (a) and native (b) species starting in 2002. Bars are one standard error of the mean (n = 6). An asterisk indicates the means in that year are significantly different.

Dicot cover was significantly (p = 0.0054) greater in burned plots than in unburned controls in 2003 after the fire (Figs 5a). The effect of the fire was gone in 2004 (Fig 5a). For dicot species, there is a significant effect of time ($p < 0.0001$) and interaction (p = 0.0142) between observation time and the fire treatment. Dicot cover increased over time in unburned control plots (Fig. 5a).

Grass (monocot) cover was significantly (p = 0.0296) less in burned plots than in unburned controls in 2003 after the fire (Figs 5b). The effect of the fire was gone in 2004 (Fig 5a). There is a significant effect of time ($p < 0.0001$) and no significant (p = 0.0548) interaction between observation time and the fire treatment. Grass cover dropped from about 72% in 2003 to about 46% in 2004 (Fig. 5b).

There was no effect of fire on species richness (Fig. 6a). Species richness increased significantly ($p < 0.0001$) over time. The index of the number of abundant species (N1) was not affected by the fire (Fig. 6b). N1 increased significantly (p = <0.0001) over time by repeated measures analysis.

The index of the number of very abundant species, N2, was significantly (p = 0.0048) greater in burned plots than in unburned controls in 2003 after the fire (Fig. 7a). The effect of the fire was gone in 2004 (Fig 7a). There was a significant effect of time ($p < 0.0001$) and interaction (p = 0.0016) between observation time and the fire treatment. N2 increased over time (Fig. 7a).

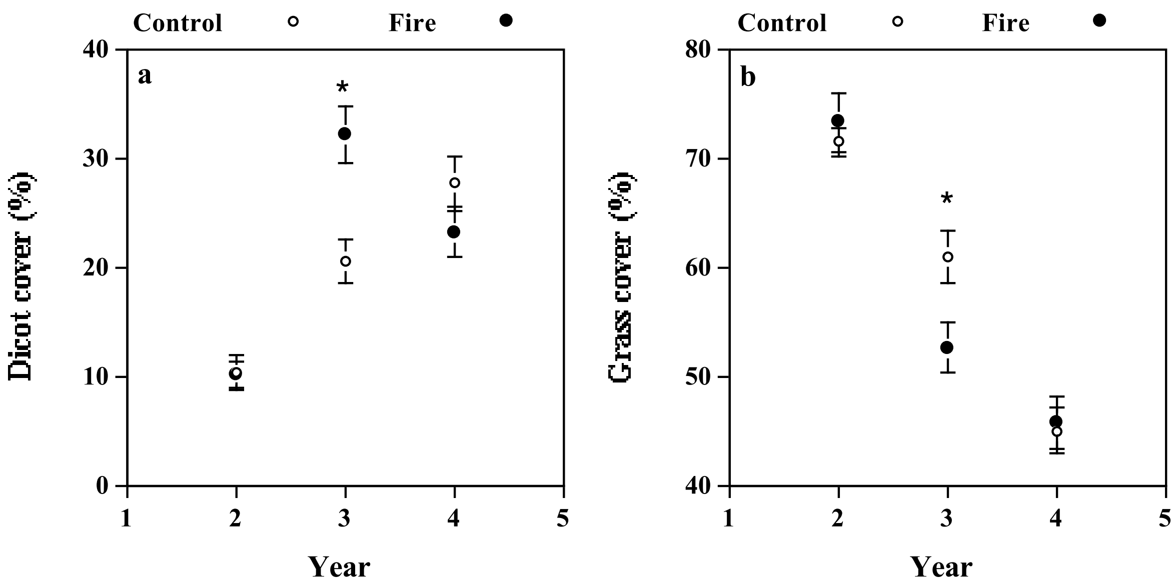

Figure 5. Percent cover of dicot (a) plus grasses (monocot) (b) species starting in 2002. Bars are one standard error of the mean (n = 6). An asterisk indicates the means in that year are significantly different.

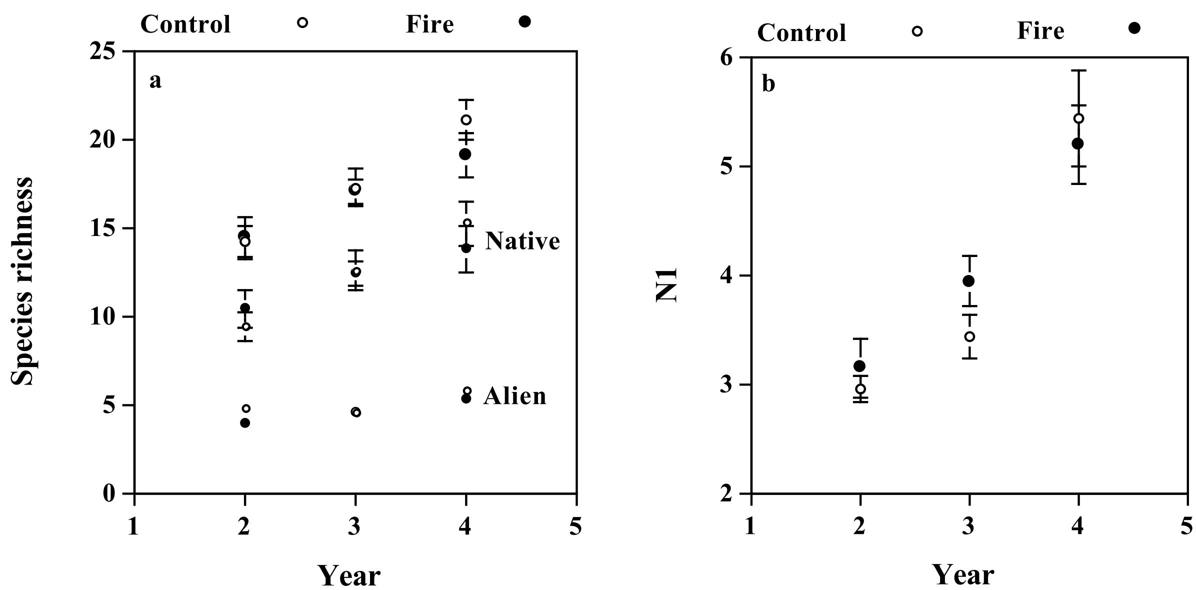

Figure 6. Species richness including that of natives and aliens (a) and N1, index of the number of abundant species (b), starting in 2002. Bars are one standard error of the mean (n = 6). An asterisk indicates the means in that year are significantly different.

Evenness was significantly ($p = 0.0054$) greater in control plots than in fire plots in 2002 before the fire (Fig 7b). In 2003 after the fire, evenness was significantly ($p = 0.0017$) greater in fire plots than in unburned controls (Figs 7b). The effect of the fire was gone in 2004 (Fig 7b). There was a significant effect of time ($p < 0.0001$) and interaction ($p = 0.0018$) between observation time and the fire treatment. Evenness increased over time (Fig. 7b).

Discussion

The main effects of the fire were an increase in the frequency of the *Descurainia* species, a reduction in *B. tectorum* cover and increases in *P. secunda* and *S. altissimum* cover. There was no effect on cover of native or alien species after the fire. We discuss these findings, how diversity indices are affected by fire, and implications for management for fire risk reduction.

Species response

Bromus tectorum cover was significantly reduced by the fire, compared with controls, in the year after the fire. The effect of the fire was gone the next year. Part of the decrease may be associated with lower litter cover after the fire. Evans and Young (1970) found density of *B. tectorum* to be much greater with litter than in bare soil in Nevada.

Bromus tectorum cover decreased over the three years in unburned plots suggesting limiting factors are controlling cover. The low cover in 2004 may be because the majority of *B. tectorum* germinated in late winter which reduces shoot size compared with plants that germinate in the fall (Mack and Pyke 1983). Water is not a strong governing factor for *B. tectorum* growth

(Link et al. 1995) unless precipitation is extremely low (Rickard and Vaughan 1988). Other factors such as available nitrogen (Cline and Rickard 1973; Link et al. 1995) and competition may be stronger factors controlling reductions in *B. tectorum* cover in this experiment.

Fires in the fall have a smaller effect on *B. tectorum* than summer burns (Klemmedson and Smith 1964; Stewart and Hull 1949).   Some summer fires are hot enough to strongly reduce *B. tectorum* seed banks so that seeding of perennials can be successful (Klemmedson and Smith 1964). Prescribed fire in the fall under the conditions of the test does not have a lasting effect on *B. tectorum*.

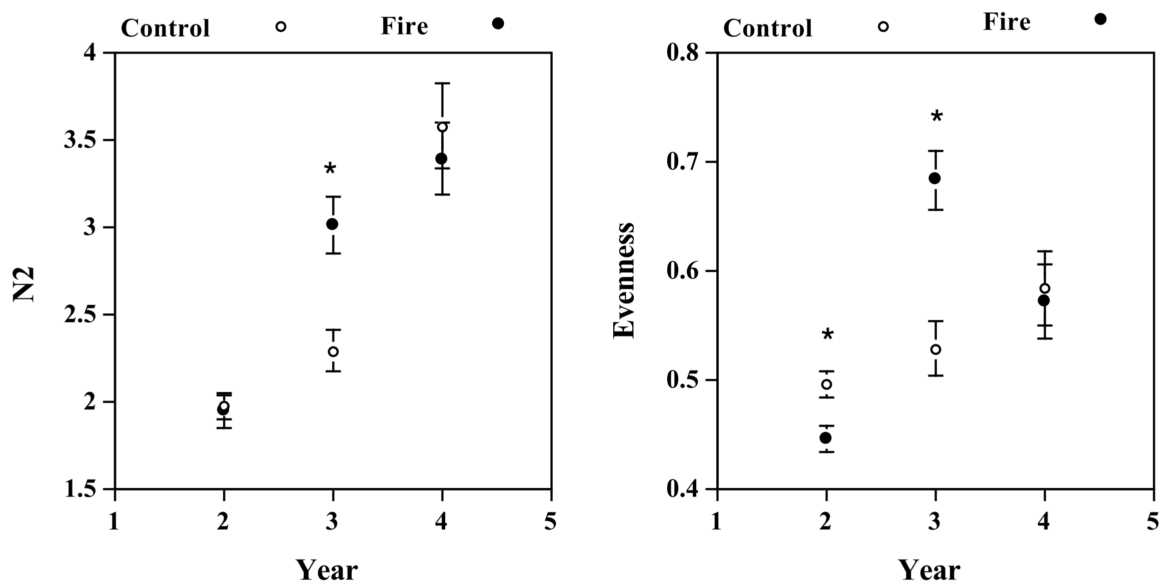

Figure 7. The index of the number of very abundant species, N2, (a) and the evenness index, E5, (b) starting in 2002. Bars are one standard error of the mean (n = 6). An asterisk indicates the means in that year are significantly different.

Poa secunda was not damaged by the fire. Its cover, after the fire, was about 6% greater in 2003 than in unburned control plots. Cover before and two years after the fire was no different than in control plots. *Poa secunda* cover is highly variable. Daubenmire (1975) found cover increased for up to four years after a burn compared with unburned controls and that variation in unburned plots was similar to natural variation found in an unburned old-field over 13 years. *Poa secunda* cover was greater one year after a summer fire than in unburned stands in Montana (Antos et al. 1983). *Poa secunda* was not affected by controlled burns at differing temperatures and season (Wright and Klemmedson 1965). Daubenmire (1975) notes that populations of *P. secunda* (a short-lived perennial) are naturally in a constant state of flux.

Sisymbrium altissimum cover greatly increased one year after the fire, but the effect of the fire was gone two years after the fire. *Sisymbrium altissimum* cover also increased significantly after a fire than in unburned controls in Montana (Antos et al. 1983). Increased *S. altissimum* cover after fire may be associated with increases in bare soil after the fire. *S. altissimum* establishment was much greater on bare soil surfaces than on litter-covered soil in Nevada (Evans and Young 1970). Two years after the fire, *S. altissimum* cover was the same as in unburned plots even though soil and litter cover were similar in the two years after the fire and much different that in unburned plots. It is possible that the amount of bare soil was greater in the late winter after the fire when *S. altissimum* germinates. Even without fire, *S. altissimum* cover varies year to year and was always below 5% in a similar ecosystem along the Snake River (Daubenmire 1975).

Amsinckia tessellata increased cover from 2002 to 2004 and was not affected by the fire. It is possible that the increase is related, in part, to reductions in *B. tectorum* cover and other species during the period reducing competition. *Amsinckia tessellata* biomass increased significantly after *Bromus* species were thinned in competition experiments in the Mojave Desert (Brooks 2000).

The *Descurainia* species became more frequent after the fire compared with unburned plots. *Descurainia pinnata* is recognized as an early seral species that does well after fire (Goodrich 1999). *Epilobium brachycarpum* became dominant after a fire in an Oregon forest, but was transient (Halpern et al. 1997). In our study, *E. brachycarpum* cover increased over the period and was not affected by the fire. Such dynamics suggest that in the study ecosystem, *E. brachycarpum* cover and frequency dynamics are not dependant on fire.

Species groups

The fire did not affect alien species cover. While there were large changes in *B. tectorum* and *S. altissimum* cover in 2003, together they were no different than in the unburned plots. The fire did not affect native species cover. Prescribed fire in the fall under the conditions of the test does not have negative affects on the native flora.

Dicot cover in burned plots was significantly greater than controls in the year after the fall fire, but the effect of the fire was gone after two years. The increase is, in large part, caused by the increase in *S. altissimum* cover. An increase in forb cover one year after the fire was observed in a similar shrub-steppe ecosystem in eastern Oregon (Young and Miller 1985). Forb cover also increased one year after a summer fire in Montana (Antos et al. 1983). Forb cover increased after a fire in a *Festuca-Stipa* grassland in Alberta (Bailey and Anderson 1978). The reduction in grass cover is, in large part, caused by the reduction in *B. tectorum* cover.

Diversity indices

Species richness was not affected by fire in this study or after fires in similar shrub-steppe communities in eastern Washington (Daubenmire 1975) and southeastern Idaho (Akinsoji 1988). Annual burns in the tallgrass prairie reduced species richness compared with unburned areas (Collins and Gibson 1990). The reduction is caused by the gradual loss of the seedbank. With less frequent burns, species richness response is highly variable (Collins and Gibson 1990). It is not likely that infrequent prescribed fall burns in conditions similar to our study will influence species richness.

The diversity index of the number of abundant species (N1) increased during the observation period. The small values in our study are similar to small values at the low end of a precipitation gradient in the Rocky Mountains and the Great Plains (Stohlgren et al. 1999). Much higher N1 values were observed at the wet end of the precipitation gradient (Stohlgren et al. 1999).

The index of the number of very abundant species (N2) increased after the fire and during the observation period. The increase in cover of *P. secunda* (Fig. 2a) and *S. altissimum* (Fig. 1b) may account for the increase after the fire given that *B. tectorum* cover remained high (about 39%) after the fire (Fig. 1a). Our values of N2 were also similar to low values associated with dry areas (Stohlgren et al. 1999).

Evenness was significantly greater in the burned plots the year after the fire which is similar to the increase in Pielou's evenness index observed by Akinsoji (1988). Fires in the shrub-steppe ecosystems similar to those in this study can reduce *B. tectorum* cover allowing more even distribution of other species. This effect is short because evenness was no longer different from that in unburned plot in the second year after the fire. After very hot fires, evenness can be reduced when a particular species takes advantage of unfilled niches and dominates (Grant and Loneragan 1999).

Management implications and Conclusions

Prescribed fire in the fall can be used to prepare surfaces for efficient application of pre-emergent herbicides in areas with 50% *B. tectorum* cover or less. The fire did not have a significant effect on native species richness or cover. A single fire is not likely to damage the ecosystem under the conditions of the test. The prescribed fire in this study reduced *B. tectorum* cover only in the year after the fire. Similarly, the fire increased cover of *S. altissimum* only in the year after the fire. Diversity effects were gone by the second year after the fire. The effect of the prescribed fire was transient for most characteristics. Only soil and litter cover remained different from unburned plots two years after the fire. Our results are limited to similar areas in the Columbia Basin. The response of other native species in different areas may be different.

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Literature cited

Akinsoji, A. 1988. Postfire vegetation dynamics in a sagebrush steppe in southeastern Idaho, USA. Vegetatio 78:151-155.

Antos, J. A., B. McCune, and C. Bara. 1983. The effect of fire on an ungrazed western Montana grassland. American Midland Naturalist 110:354-364.

Bailey, A. W., and M. L. Anderson. 1978. Prescribed burning of a Festuca-Stipa grassland. Journal of Range Management 31:446-449.

Billings, W. D. 1990. *Bromus tectorum*, a biotic cause of ecosystem impoverishment in the Great Basin. Pp. 301-322 *in* G. M. Woodwell, ed. The earth in transition: patterns and processes of biotic impoverishment. Cambridge University Press, Cambridge.

Bond, W. J., and B. W. v. Wilgen. 1996. Fire and Plants. Chapman and Hall, London.

Bonham, C. D. 1989. Measurements for Terrestrial Vegetation. John Wiley & Sons, New York.
Bradstock, R. A., M. Bedward, B. J. Kenny, and J. Scott. 1998. Spatially-explicit simulation of the effect of prescibed burning on fire regimes and plant extinctions in shrublands typical of south-eastern Australia. Biological Conservation 86:83-95.
Brooks, M. L. 2000. Competition between alien annual grasses and native annual plants in the Mojave Desert. American Midland Naturalist 144:92-108.
Cline, J. F., and W. H. Rickard. 1973. Herbage yields in relation to soil water and assimilated nitrogen. Journal of Range Management 26:296-298.
Collins, S. L., and D. J. Gibson. 1990. Effects of fire on community structure in tallgrass and mixed-grass prairie. Pp. 81-98 *in* S. L. Collins and L. L. Wallace, eds. Fire in North American tallgrass prairies. University of Oklahoma, Norman.
D'Antonio, C. M., and P. M. Vitousek. 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global change. Annu. Rev. Ecol. Syst 23:63-87.
Daubenmire, R. 1970. Steppe vegetation of Washington, Pullman, Washington.
Daubenmire, R. 1975. Plant succession on abandoned fields, and fire influences, in a steppe area in southeastern Washington. Northwest Science 49:36-48.
Elmore, A. J., J. F. Mustard, and S. J. Manning. 2003. Regional patterns of plant community response to changes in water: Owens Valley, California. Ecological Applications 13:443-460.
Evans, R. A., and J. A. Young. 1970. Plant litter and establishment of alien weed species in rangeland communities. Journal of Range Management 18:697-703.
Goodrich, S. 1999. Multiple use management based on diversity of capabilities and values within pinyon-juniper woodlands. USDA, Forest Service, Rocky Mountain Research Station, RMRS-P-9:164-171
Grant, C. D., and W. A. Loneragan. 1999. The effects of burning on the understorey composition of 11-13 year-old rehabilitated bauxite mines in Western Australia. Plant Ecology 145:291-305.
Halpern, C. B., J. A. Antos, M. A. Geyer, and A. M. Olson. 1997. Species replacement during early secondary succession: the abrupt decline of a winter annual. Ecology 78:621-631.
Hitchcock, C. L., and A. Cronquist. 1976. Flora of the Pacific Northwest. University of Washington Press, Seattle.
Hurlbert, S. H. 1984. Pseudoreplication and the design of ecological field experiments. Ecological Monographs 54:197-211.
Klemmedson, J. O., and J. G. Smith. 1964. Cheatgrass (*Bromus tectorum* L.). The Botanical Review 30:226-262.
Knapp, P. A. 1998. Spatio-temporal patterns of large grassland fires in the Intermountain West, U.S.A. Global Ecology and Biogeography Letters 7:259-272.
Knick, S. T., and J. T. Rotenberry. 1997. Landscape characteristics of disturbed shrubsteppe habitats in southwestern Idaho (U.S.A.). Landscape Ecology 12:287-297.
Link, S. O., J. H. Bolton, M. E. Thiede, and W. H. Rickard. 1995. Responses of downy brome to nitrogen and water. Journal of Range Management 48:290-297.

Link, S. O., C. W. Keeler, R. W. Hill, and E. Hagen. 2005. *Bromus tectorum* cover mapping and fire risk. International Journal of Wildland Fire (submitted 12-22-04)

Link, S. O., W. H. Mast, R. W. Hill, and M. Pellent. 2004. Restoration Encyclopedia: Shrub-steppe Restoration. In: Proceedings of the16th Int'l Conference, Society for Ecological Restoration, Victoria, BC

Ludwig, J. A., and J. F. Reynolds. 1988. Statistical ecology: a primer on methods and computing. John Wiley, New York.

Mack, R. N. 1981. Invasion of *Bromus tectorum* L. into western North America: an ecological chronical. Agro-Ecosystems 7:154-165.

Mack, R. N., and D. A. Pyke. 1983. The demography of Bromus tectorum: variation in time and space. Journal of Ecology 71:69-93.

Monsen, S. B. 1994. The competitive influences of cheatgrass (*Bromus tectorum*) on site restoration. Proceedings - ecology and management of annual rangelands,

Pellant, M. 2002. Cheatgrass: Invasion, Occurrence, biological/competititve features and control measures. Restoration and Management of Sagebrush/Grass Communities Workshop, Elko, NV 29-33.

Rickard, W. H., and B. E. Vaughan. 1988. Plant community characteristics and responses. Pp. 109-179 *in* W. H. Rickard, L. E. Rogers, B. E. Vaughan and S. F. Liebetrau, eds. Shrub-steppe: balance and change in a semi-arid terrestrial ecosystem. Elsevier, Amsterdam.

Steele, R. B. D., and J. H. Torrie. 1960. Principals and Procedures of Statistics. McGraw-Hill, New York.

Stewart, G., and A. C. Hull. 1949. Cheatgrass (*Bromus tectorum* L.)-An ecologic intruder in southern Idaho. Ecology 30:58-74.

Stohlgren, T. J., L. D. Schell, and B. V. Heuvel. 1999. How grazing and soil quality affect native and exotic plant diversity in Rocky Mountain grasslands. Ecological Applications 9:45-64.

Whelan, R. J. 1995. The ecology of fire. Cambridge University Press, Cambridge.

Whisenant, S. G. 1990. Changing fire frequencies on Idaho's Snake River plains: ecological and management implication. USDA Forest Service Intermountain Research Station General Technical Report: INT-276:4-10

Wright, H. A., and J. O. Klemmedson. 1965. Effect of fire on bunchgrasses of the sagebrush-grass region in southern Idaho. Ecology 46:680-688.

Young, R. P., and R. F. Miller. 1985. Response of Sitanion hystrix (Nutt.) J. G. to prescribed burning. American Midland Naturalist 113:182-187.