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Fire and False Brome: How Do Prescribed Fire and Invasive *Brachypodium sylvaticum* Affect Each Other?

Lauren P. Poulos and Bitty A. Roy*

Brachypodium sylvaticum, a shade-tolerant, forest dwelling, and aggressive invasive grass native to Eurasia, is a noxious weed in California, Oregon, and Washington. This species could cause ecosystem collapse by altering forest fire regimes. To examine interactions with fire, we divided two Willamette National Forest sites into eight units and randomly selected half for treatment with prescribed fire in spring 2011. We assessed the effect of *B. sylvaticum* on fire (severity and intensity) as well as the effect of fire on *B. sylvaticum* (cover, seedling emergence, and dispersal). We found that *B. sylvaticum* cover decreased fire severity but had no effect on intensity. Furthermore, fire severity influenced *B. sylvaticum* cover; in areas receiving low-severity fire, the grass increased from 21 ± 15.05 to 34 ± 15.81%, but in areas of high-severity fire, cover remained consistently around 0% (0 ± 0% cover in yr 1 to 0.2 5± 0.25% in yr 3). In the field, prescribed fire decreased seedling emergence by 32% compared to controls, but not in an associated greenhouse experiment. However, in the greenhouse, severely burned plots had zero emergence, compared to 0.29 ± 0.14 seedlings low-severity m^{-2} plot. Fire severity also influenced dispersal in the field; we monitored plots with < 0.5% cover *B. sylvaticum* initially; when these plots experienced low severity fire, they had greater *B. sylvaticum* cover (increasing 1,200%), suggesting increased dispersal with less severe fires. High-severity dispersal plots did not experience increased cover. High severity fires have the potential to control the grass, but low-severity fires will likely increase its cover.

Nomenclature: False brome, *Brachypodium sylvaticum* (Huds.) Beauv.

Key words: Conifer forest, Douglas fir, fire intensity, fire severity, invasive grass, prescribed fire, shade tolerance.

One current ecological challenge is the remediation of nonnative invasive plant species before they irreparably transform ecosystems by altering disturbance regimes (D'Antonio and Vitousek 1992), available abiotic resources, native vegetation structure, or patterns of native establishment and recruitment (Gordon 1998; Vitousek et al. 1996). The nonnative grass *Brachypodium sylvaticum* (Huds.) P. Beauv., commonly known as false brome, is an example of a species that could potentially cause major ecosystem change through each of these different pathways.

The native range of *B. sylvaticum* is in temperate forests throughout most of Eurasia, including Europe (from Scandinavia to Spain), Russia, China, Japan, India, and Indonesia, as well as Lebanon, Syria, Iran, Algeria, and Eritrea (Roy 2010). It is a perennial, wind-pollinated grass (Rosenthal et al. 2008) that primarily grows in forests where it tolerates a range of light conditions from deep shade to open canopy (Corney et al. 2008; Holmes et al. 2010; Hrusa 2003; Murchie and Horton 2002; Palo et al. 2008; Parks et al. 2005). Because shade tolerant taxa in general are relatively unusual, this characteristic might give *B. sylvaticum* a competitive advantage (Martin et al. 2009; Sutherland 2004) because it has the ability to utilize and exploit a variety of environments.

First found in North America in Oregon in 1939 (Chambers 1966), it is now considered extremely invasive and has been declared a noxious weed in California, Oregon, and Washington (CDFA 2009; NWCB 2009; ODA 2009). A rapid seed disperser (Petersen and Philipp 2001), the lengthy awns of this grass work into animal fur (Heinken and Raudnitschka 2002), which facilitates transportation. The range of this species is rapidly expanding in the United States; it has been noted as far east as Missouri, New York, and Virginia (Roy 2010). False brome is associated with logging practices, because it is transported on equipment and crews (Fletcher 2009; USDA Forest Service et al. 2009). It is also moved by human recreational activities and by rivers and streams (Roy 2010). Similar to many invasive species, in its

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Management Implications

In the conifer forests of the Pacific Northwest, the nonnative grass *Brachypodium sylvaticum* (false brome) is forming "lawns" in forests that have never had continuous grass cover. Can prescribed fire be used to control this invasive grass that decreases tree germination and growth? Our results suggest that if a uniform, high-severity fire can be achieved and sustained, this method of treatment can be used to control *B. sylvaticum*. Unfortunately, fires do not often burn uniformly, but instead usually create mosaics of varying severity across a landscape, thus rendering this objective challenging. The stakes are high given that we showed that low-severity burns actually increase the seedling emergence, apparent dispersal, and cover of *B. sylvaticum*. Therefore, it is difficult to prescribe fire to control the grass. However, if prescribed burns are being used anyway for other reasons, or if natural fires occur, then there is opportunity for controlling *B. sylvaticum* by adding in on-the-ground control methods (such as herbicide or mechanical removal) in the less-severely burned zones.

nonnative range it is highly correlated with disturbed areas. For example, a recent study found that invasions were concentrated within 30 m (98.4 ft) of human-use corridors such as power lines and trails (Holmes et al 2010).

Brachypodium sylvaticum grows into thick, lawn-like monocultures, and invasive grasses in general have been shown to crowd out native plants and decrease recruitment of conifers (Kruse et al. 2004; Lehmkuhl 2002; Powell et al. 1994). Because forests in western Oregon have not evolved with an understory of dense grass, this change in ecosystem structure could have a variety of unforeseen consequences. Besides reducing natural native tree regeneration, the loss of biodiversity is already resulting in a ''trophic cascade'' (Zuefle et al. 2008) in which species in other trophic levels are negatively affected. Population levels of an endangered butterfly are decreasing because of incursions of this grass (Severns and Warren 2008).

Brachypodium sylvaticum is changing forest fuel structure with the addition of thick thatch layers made up of each year's senesced growth, which could affect the fire regime, defined as the characteristic fire frequency, intensity, severity, and spatial/temporal scales of a given area (Gill 1975; Pausas and Keeley 2009). We define fire intensity as the measure of energy output by a fire (Keeley 2009), or for our purposes, fire temperature. Fire severity describes the actual ecosystem consequences from fire (e.g., changes to vegetation, percent scorch, fuel consumed) and is not to be confused with intensity (Keeley 2009). Because *B. sylvaticum* is a perennial grass, thick layers of dry thatch accumulate each season, and with a large enough infestation, this accumulated dry, flashy fuel could be a driver for fires of higher severity and/or intensity.

A nonnative grass introduced to an area is sometimes enough to set in motion the ''grass–fire cycle'' (D'Antonio and Vitousek 1992). Often times, invasion also interacts with human-mediated land-use change, creating a positive feedback loop because more disturbed areas are more at risk for invasion (DiTomaso et al. 2006). Grasses increase the probability of fire (Keeley 2006), but they typically create lower-intensity fires than dense shrub understories (Zschaechner 1985), which can have the consequence of preserving the seed survival rate due to the lower energy output (i.e., lower temperature) of grass fires (Keeley 2006). With an increased accumulation of fine, flashy fuels, grasses can increase fire frequency and area burned, and then outcompete native species for space and resources postburn. As the invading species increases in cover, so does the strength of the grass–fire cycle, and essentially, it becomes self-propagating unless something intervenes to stop its progress. In other words, *B. sylvaticum* might alter the ecosystem in ways that increase the likelihood of its own success.

Is *Brachypodium sylvaticum* a species capable of initiating a grass–fire cycle? If so, the results could have tremendous implications for the forest ecosystems throughout the western United States. Not only are there potential severe ecosystem consequences, but also financial ones. This grass is invading the western side of the Cascade mountains, an area that is highly valued for its timber production. It is imperative that we fully understand how this particular grass interacts with fire in these forests. Employing the use of prescribed fire, we focused on two questions: (1) Does *B. sylvaticum* affect fire intensity and/or severity? and (2) Does treatment with fire affect *B. sylvaticum* cover, seedling emergence, or dispersal?

Materials and Methods

Study Sites. The Cascade Range of the western United States is known for its steep, volcanic terrain with mean annual precipitation exceeding 250 cm (98.4 in), falling from October through April as rain in lower elevations and snow in higher areas (Cissel et al. 1999). The Willamette National Forest (WNF) stretches along the western side of these slopes in Oregon, and has a historical fire patch size of about 10 to 160 ha (25 to 395 ac), excluding very large, stand-replacing fires that have infrequently occurred every 200 to 415 yr and created much larger fire patches, some thousands of ha in size (Cissel et al. 1999).

Two sites within the WNF were selected for study due to pre-existing patchy infestations of *B. sylvaticum* (Site 1: 44°09′40.30″N, 122°02′48.60″W; Site 2: 44°9′32.20″N, 122°3′23.98″W). Each site is 760 m in elevation and dominated by Douglas fir [*Pseudotsuga menziesii* (Mirb.) Franco] and western hemlock [*Tsuga heterophylla* (Raf.) Sarg.] coniferous forest. Site 1 is 9.7 ha and Site 2 is 24.7 ha, totaling 34.4 ha. Both sites were divided into eight equal-sized subunits with half of the subunits in each site randomly selected for treatment with prescribed fire and the other half left as controls. To monitor cover of

Table 1. Environmental data collected by site, transect, and plot. Canopy cover measured in percent cover by plot (means ± standard error [SE]).

	Site 1	Site 2
Data by site		
Relative humidity (%)	46	36
Temperature (°C)	22	27
Data by transect		
Slope, range (°)	(1–37)	0
Slope, (°) mean ± SE	18 ± 4.80	0 ± 0
Data by plot		
Preburn canopy (%)	26 ± 1.54	22 ± 2.22
Postburn canopy (%)	26 ± 1.59	21 ± 2.23

B. sylvaticum pre- and postburn, 16 1-m^2 plots containing *B. sylvaticum* in each site (two plots randomly located in each subunit) were established. To determine whether fire influences dispersal, we set up 32 1-m^2 (3.3 ft^2) plots without *B. sylvaticum* (cover varying between 0 and $< 0.5\%$, or one nonreproductive seedling), positioned perpendicular to the *B. sylvaticum* plots and 10 m away from the invading front. In these *Brachypodium*-free plots, we measured *B. sylvaticum* cover over time to study the rate of spread to new areas. Each site thus had eight burned and eight unburned 1-m^2 *B. sylvaticum* plots, as well as 16 burned and 16 unburned 1-m^2 dispersal plots for a total of 96 plots (Supplemental Figure S1; http://dx.doi.org/10.1614/IPSM-D-14-00024.S1).

To be sure the treatments and controls were similar before starting and to increase our ability to detect the influence of the fire treatment, we collected both pre- and postburn data. The prescribed burn was applied using drip torches, which apply a flaming mixture of diesel fuel and gasoline as a drizzle when tipped toward the ground, allowing for fine-scale control of where the flame is applied and in what quantities. The fires were set on June 22, 2011 to Site 1 and July 10 2011 to Site 2. Although about 2 wk separated the fires, the sites were similar at the time of ignition in terms of fire readiness (the second site was more densely vegetated, both in the understory as well as canopy, and thus required more time to lose fuel moisture and become acceptably flammable). Site humidity and temperatures on the day the fires were set, as well as slope measurements and canopy cover by plot can be found in Table 1.

Preburn measurements were taken in fall 2010 (vegetation) and spring 2011 (fuels). All postburn measurements that were the most sensitive to weather influences (for example, burn severity is partially assessed by levels of ash on the ground) were gathered the day each prescribed burn was applied. General vegetation monitoring was conducted in the preburn fall 2010 and postburn falls 2011 and 2012, whereas seedling counts were taken in spring and fall 2012.

Percent cover of *B. sylvaticum*, other vegetation, rock/soil (bare ground), and total wood (dead and fallen woody debris) was estimated visually with a 1-m^2 quadrat, taking into account the area from the ground level to a height of about 3 m. (Supplemental Figure S2; http://dx.doi.org/10.1614/IPSM-D-14-00024.S1). Percent cover of total woody debris was also broken down into different fuel classifications based on diameter of the woody fuel and estimated pre- and postburn 2011 (Supplemental Figure S3; http://dx.doi.org/10.1614/IPSM-D-14-00024.S1).

Brachypodium sylvaticum seedling counts were taken postburn in spring and fall 2012. These measurements were used to address the question of whether fire affected the next year's seed bank (control vs. burned plots). As a perennial bunchgrass, *B. sylvaticum* exhibits senesced and dried material from the previous year, which marks the outer perimeter of the plant. Seedlings are obvious because, as opposed to the larger bunchgrass parent, they are generally composed of only one or two blades of grass, sometimes with a visible attached seed, and were counted only when it was obvious that they stood apart from the parent.

Due to extremely high densities of *B. sylvaticum* seedlings and established individuals in many plots, field counts were taken only at the quarter-plot scale. Only nonreproductive individuals less than 7 to 10 cm in height and with three or fewer leaves (and sometimes with seed visible) were counted as seedlings. Counts were consistently taken from the northeast corner of each plot.

Fire Severity. Fire severity was measured as an index on the day of the fire for each burned plot and was assessed using a combination of substrate and vegetation characteristics (e.g., levels of char or scorch, amount of woody debris or vegetation consumed, ash levels/colors) (USDI National Parks Service 2003). The scale is ranked in descending order with a category 5 severity as the lowest (unburned) and category 1 as highest severity (heavily burned). Those plots with a classification of 5 had fire applied but

displayed very little, if any visible consequences of burning. The severity scale is illustrated with our plots in the Supplemental Figure S4 (http://dx.doi.org/10.1614/IPSM-D-14-00024.S1).

Fire Intensity. Fire intensity was measured using the maximum observed plot temperature during fire treatment and was captured with temperature-sensitive paints (Iverson et al. 2004). Because 60 C is the the lowest temperature capable of causing lethal plant tissue damage (Wally et al. 2009), and forest fires exhibit high temperatures ranging from 800 to 900 C (Wally et al. 2009), we chose seven different paints, each sensitive to a specific temperature: 79 C, 149 C, 253 C, 343 C, 454 C, 649 C, 816 C (OmegaLaq®, Omega). Paints were applied to copper garden tags in descending temperature order, with each paint present on every tag, and with a blank tag secured to the painted tag to protection the paint from the elements. Tags were secured to a stake and arranged in pairs: one at ground level and the other suspended 20 cm. There were two replicates per plot, for a total of four tag readings, which were then averaged together for one temperature reading per plot (Supplemental Figure S5; http://dx.doi.org/10.1614/IPSM-D-14-00024.S1). Stakes were placed in both the northwest and southeast plot quarters. Tags were set up about 2 wk prior to the fire being set.

Environmental Factors. Ambient temperature and relative humidity at time of the burn was recorded for each site and communicated to us by the U.S. Forest Service fire personnel (Table 1). Canopy cover was measured using a convex spherical densiometer pre- and postburn for each of the four sides of the quadrat, and the readings were averaged for an overall plot measurement (Table 1). Slope was measured once, preburn, with an inclinometer (Table 1).

In the event of an unforeseen weather occurence, such as a drought, resulting in little germination in the field, we also conducted a greenhouse experiment. Four soil cores (5 cm in diam by 5 to 7 cm deep) were taken within 0.5 m of every plot postfire and composited. The soil for each plot was added to a sand and sphagnum mixture in a 1.5 : 4 ratio (soil : sand/sphagnum) and spread into sterile 25 by 50 cm trays. These were randomly arranged in the greenhouse, watered every other day, and exposed to 16 h of light d^{-1} for 1 wk, followed by 6 wk of 12-hr d. Only *B. sylvaticum* and other graminoids were identified and counted.

Statistical Analysis

To gain an understanding of which aspects of the fire, if any, were important, we ran independent analyses for fire intensity and severity. It is necessary to analyze both intensity and severity because it is completely possible for a high-intensity (hot) grass fire to burn rapidly, and thus do little damage (severity). In other words, a range of fire intensities can produce a single severity (Wade 2013).

Analysis of overall fire effects included all plots, control and burn. Because severity and intensity metrics only occurred in burned units, we ran these analyses only on burn plots and excluded controls. All percent cover and proportion data were logit transformed to meet the normality assumptions of the analyses (Warton and Hui 2011). JMP® Pro 9.0.2 statistical software (SAS 2010) was used for statistical analyses, and Microsoft Excel (2010) was used for data keeping.

To address our study question of whether *B. sylvaticum* alters fire intensity and/or severity, we applied a backwards stepwise regression using the lowest Bayesian information criterion (BIC) value in order to understand which variables were influencing fire behavior, because fire can be influenced by an array of environmental variables, any of which could potentially be driving the system. Analyzing only the burn plots, we first tested for multicollinearity of variables (slope, preburn canopy cover, *B. sylvaticum* cover, other vegetation cover, and cover of each of the fuels classes), which was not an issue (the highest variance inflation factor [VIF] was 2.7). Fire severity was treated as ordinal data, and intensity as continuous data. To reduce the number of variables included in the model, we used site as a proxy for environmental variables such as relative humidity and ambient temperature at the time of the burn.

To address our second question of whether prescribed fire affects cover of *B. sylvaticum*, we employed a repeated measures analysis of variance (ANOVA) with site as a random factor. Comparing each plot's percent cover of *B. sylvaticum* over time, we had three dependent measurements of cover (one preburn and two postburn). We looked at differences in the means over time using the measured explanatory fire variables (i.e., severity and intensity).

To assess the consequences of prescribed fire on *B. sylvaticum* seedling emergence in the field, we log-transformed the spring 2012 seedling counts and applied a Type I, or sequential test, to control for the variation contributed by preburn cover of *B. sylvaticum* in each plot. In this way, we were able to control for the influence of previously established individuals contributing to the seedling counts and strictly assess the influence of fire on any resulting seedling emergence. We regressed the log-transformed seedling counts in a sequential test against the logit-transformed preburn *B. sylvaticum* cover, treatment (control vs. burn), and site. We treated site as a random effect in a restricted maximum likelihood (REML) model so that our results would be generally applicable instead of only pertinent to our specific study sites. We repeated the sequential test described for the field counts above again for the greenhouse data. Log transformation did not produce

normal residuals of the greenhouse seedling counts, so we used a Box Cox power transformation.

For the statistical model testing the effect of prescribed fire on *B. sylvaticum* dispersal, only the dispersal plots were analyzed (i.e., only the plots that contained < 0.5% cover of *B. sylvaticum* at the inception of the study), and a repeated measures ANOVA was applied to look at the change in *B. sylvaticum's* cover over time and with and without the prescribed burn.

Results and Discussion

Does *B. sylvaticum* Affect Fire Severity and/or Intensity? Only two variables significantly influenced fire severity: *B. sylvaticum* cover ($F_{1,43}$ = 6.37, P = 0.0154) and 100-h fuels ($F_{1,43}$ = 7.25, P = 0.0059). *Brachypodium sylvaticum* cover decreased fire severity; plots containing greater cover of the grass experienced lower severity burns (Figure 1A). There was no significant relationship between *B. sylvaticum* cover and fire intensity ($F_{1, 43}$ = 0.74, *P* = 0.39) (Figure 1B); the variables that significantly drove intensity were site ($F_{1,43}$ = 11.96, P = 0.001) and preburn 10-h fuels ($F_{1,43}$ = 0.66, P = 0.02), which is woody debris 0.6 to 2.6 cm in diam.

Brachypodium sylvaticum stays green through the dry summers of the Pacific Northwest. For this reason, we predicted that it could have a dampening effect on fire, which is supported by our findings concerning fire severity. Plots with high *B. sylvaticum* cover had lower-severity fires (Figure 1A); other environmental variables were more influential for fire intensity.

Does Prescribed Fire Affect *B. sylvaticum*? When looking at burned vs. control plots, prescribed fire did not affect cover of *B. sylvaticum* ($F_{1,62}$ = 0.66, P = 0.42). When we examined only the burned plots, we found no significant relationship between intensity of fire and *B. sylvaticum* cover ($F_{1,46}$ = 3.64, P = 0.06) (Table 2A); intensity tended to be negatively associated with *B. sylvaticum* cover.

Severity of fire on *B. sylvaticum* cover appears to be the true driver of change in this temperate, Douglas fir- and western hemlock-dominated coniferous forest system ($F_{4,43}$ = 5.01, P = 0.002) (Table 2B and Figure 2). There was no obvious change in percent cover until 2012 when those plots which burned with the lowest severity (level 5) showed a marked increase in cover from 21 ± 15.05 to 34 ± 15.81%, and those that burned with the highest severity fire (level 1) did not exhibited much of a change in cover of the grass, going from an average of 0 ± 0% cover to 0.25 ± 0.25% cover (Figure 2).

Overall, there were no differences in *B. syvlaticum* cover between fire and control plots. However, breaking down the fire treatment into intensity and severity, there was a trend toward severity but not intensity influencing

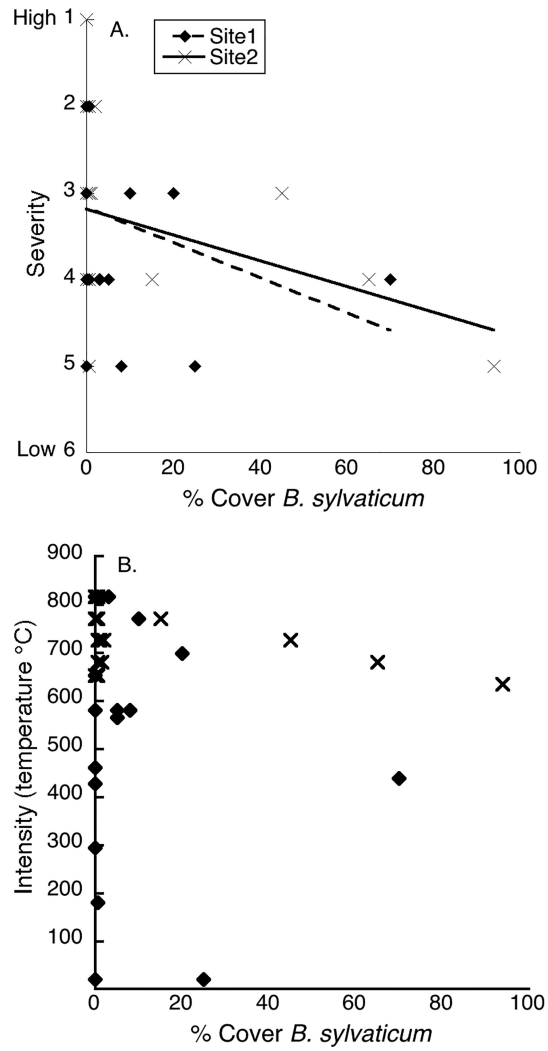

Figure 1. Preburn *B. sylvaticum* (false brome) cover on (A) fire severity and (B) fire intensity. Preburn *B. sylvaticum* cover had a significant effect on fire severity but not fire intensity; to improve readability, percent cover is shown in both graphs, but the analysis was done on logit-transformed data. As illustrated in Panel A, plots with high cover of the grass experienced lower severity burns ($F_{1,43}$ = 6.37, P = 0.0154). In Panel B, *B. sylvaticum* cover did not have a significant effect on fire intensity ($F_{1,43}$ = 0.74, P = 0.39).

B. sylvaticum cover. It is likely that severity was the more appropriate measure for this analysis because it captures the actual ecosystem effects after fire. Just because a fire burned with high intensity (higher temperatures) does not necessitate there being any damage to target vegetation; the flaming front might have passed so quickly that there was little disturbance to vegetation or soil, for example.

In other studies involving *B. sylvaticum*, fire was observed to reduce *B. sylvaticum* cover or was negatively associated with *B. sylvaticum* cover (Arévalo et al. 2001; DiTomaso et al. 1999, 2006; Safaian et al. 2005).

Table 2. Fire affects on *Brachypodium sylvaticum* (false brome). The results are shown from three separate repeated-measures ANOVA analyses, each with site treated as a random factor. Panels A (intensity) and B (severity) illustrate the differences in the means over time for percent cover of *B. sylvaticum* by plot in response to the measured explanatory fire variables. Panel C illustrates the differences in means over time for seedling emergence by plot in response to the measured fire variable, severity.[a]

Factor	MS	F	df	P
A. Fire intensity affects *B. sylvaticum* cover				
Between				
All	0.08	3.64	1,46	0.0627
Intercept	0.08	3.89	1,46	0.0547
Intensity	0.08	3.64	1,46	0.0627
Within				
All	0.16	3.56	2,45	0.0368[b]
Time	0.09	2.00	2,45	0.1475
Time by intensity	0.16	3.56	2,45	0.0368[b]
B. Fire severity affects *B. sylvaticum* cover				
Between				
All	0.47	5.01	4,43	0.0021[b]
Intercept	0.42	18.17	1,43	0.0001[b]
Severity	0.47	5.01	4,43	0.0021[b]
Within				
All	0.49	4.54	8,84	0.0001[b]
Time	0.03	0.61	2,42	0.5487
Time by severity	0.49	4.54	8,84	0.0001[b]
C. Fire severity affects *B. sylvaticum* seedling emergence				
Between				
All	0.81	5.50	4,27	0.0023[b]
Intercept	4.42	119.40	1,27	$<$ 0.0001[b]
Severity	0.81	5.50	4,27	0.0023[b]
Within				
All	0.49	2.80	8,52	0.0118[b]
Time	0.06	0.82	2,26	0.4525
Time by severity	0.49	2.80	8,52	0.0118[b]

[a] Abbreviations: MS, mean squares; F, F statistic; df, degrees of freedom; P,P value; X, interaction term.

[b] significant at 0.05.

Additionally, *B. sylvaticum* shoot biomass was observed to be lower when grown in high–severity-burn soil as compared to low-severity soil (Hebel et al. 2009). Other researchers have observed a decrease in plant biomass with species other than *B. sylvaticum* with increasing burn severity for up to 2 yr after fire treatment (Feller 1996), perhaps as a result of loss of soil organic matter.

Because high-severity fires reduce *B. sylvaticum* cover, it might be beneficial to utilize repeated high-severity fires to eradicate, or at least slow, the invasion. However, because high-severity prescribed fire could run a greater risk of starting unintentional wildfires, the potential cost in lost timber sales or resources during suppression might not outweigh the benefit for managing agencies. Our prescribed burns were implemented at the end of a very wet and late spring, and perhaps because of this, we were unable to get high-severity fire in areas with high *B. sylvaticum* cover due to the dampening from the live green biomass outweighing the flammability of the dead growth from the previous year. If a late-summer or -fall burn were applied, the results might show that higher-severity fires could be attained, thus lowering *B. sylvaticum* cover; however, the risk of spot fires or wildfires would potentially increase as well. Perhaps a series of small-area, high-severity burns would be more manageable logistically.

Unfortunately, low-severity fires appear to increase *B. sylvaticum* cover, which suggests that *B. sylvaticum* might be an invasive grass species capable of instigating a grass–fire cycle. Areas of high grass cover burn at higher severity, which results in even higher densities, and the grass could potentially outcompete native species for space and resources. Likely, there is a threshold level of density at which this cycle is set in motion, after which it becomes self-propagating, with more grass leading to more fire and

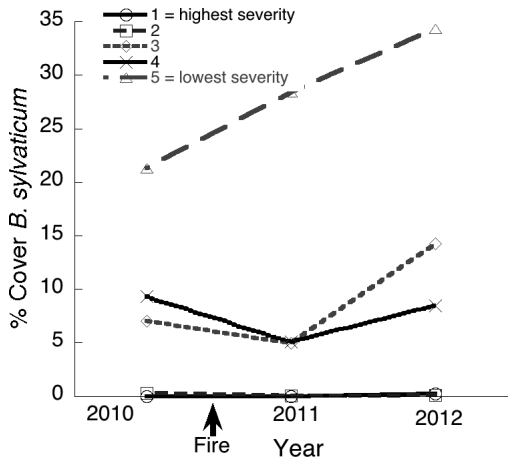

Figure 2. Severity of fire and *B. sylvaticum* cover. Norms of reaction graph showing the interactions among year, fire severity, and cover of *B. sylvaticum* in the main experiment. The points are averages; they are connected by lines to illustrate the lack of parallel lines, indicating a statistical interaction. Plots that burned with the lowest severity showed an increase in *B. sylvaticum* cover ($F_{4,43}$ = 5.01, P = 0.002, repeated measures ANOVA). Note that the fire was initiated between 2010 and 2011.

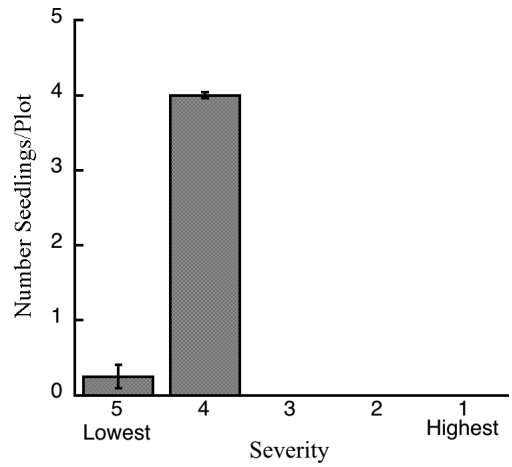

Figure 3. Greenhouse seedling emergence data. High-severity plots had significantly lower seedling emergence rates ($F_{1,48}$ = 5.05, P = 0.03, R^2 = 0.13). Means ± standard error (SE).

more grass. The grass–fire cycle is most often thought of as a mechanism for converting shrublands to grasslands; however, the original description of the cycle (D'Antonio and Vitousek 1992) also discussed the possibility of woodlands and logged sites being vulnerable to the grass–fire cycle.

Analyzing the seedling counts taken in the forest the year following treatment, prescribed fire had an overall negative influence on *B. sylvaticum* seedling emergence in the field ($F_{1,96}$ = 7.54, P = 0.007, R^2 = 0.46), decreasing emergence by 32% in burned plots when compared to control (6.88 ± 3.04 seedlings burn plot^{-1} vs. 9.04 ± 2.61 seedlings control plot^{-1}). Site explained only 6.3% of the variance in the data. Looking more closely at the fire effects, we saw that severity of fire ($F_{4,48}$ = 1.62, P = 0.19, R^2 = 0.49) and intensity of fire ($F_{1,48}$ = 2.11, P = 0.16, R^2 = 0.47) separately were not significant influences. In the greenhouse experiment, we did not see a significant influence of treatment (fire vs. controls) ($F_{1,96}$ = 1.00, P = 0.32, R^2 = 0.05) or intensity of fire ($F_{1,48}$ = 0.76, P = 0.39, R^2 = 0.06) on seedling emergence. We did, however, observe a significant effect for fire severity ($F_{1,48}$ = 5.05, P = 0.03, R^2 = 0.13) with no emergence in high severity plots, compared to 0.29 ± 0.14 seedlings low-severity m^{-2} plot; higher-severity fire led to lower seedling emergence (Figure 3).

Prescribed fire signficantly decreased seedling emergence in the field. When controlling invasive species with fire, it is imperative to kill the target plants before their seeds become viable (DiTomaso et al. 1999) or critically damage the seeds before dispersal can take place (Allen et al. 1995; Menke 1992;). Studies of most perennial *Poa* species have found that burning in mid- to late spring is most effective (Becker 1989; Curtis and Partch 1948; Engle and Bultsma 1984), the same timing as we used. However, in the Pacific Northwest, a fall burn might be more effective, because it would be drier, allowing for a more homogenous high-severity fire.

We measured dispersal by setting up plots that were 10 m from the invading front that had < 0.5% false brome cover, then tracked *B. sylvaticum* cover over time. When we compared fire vs. control plots with a repeated measures ANOVA, there was no significant effect on dispersal ($F_{1,62}$ = 0.66, P = 0.42), but when looking specifically at the severity of the fire, we found that it significantly affected dispersal ($F_{4,27}$ = 5.50, P = 0.002) (Table 2C and Figure 4). The dispersal plots with the highest-severity fires gained fewer plants over time (no average gain of cover), whereas the plots with the lowest-severity fires gained the most plants over time (increasing by more than 1,200% from an average of 0.32 ± 0.07% cover plot^{-1} to an average of 3.89 ± 1.78% cover). Intensity of fire did not have a significant effect on dispersal ($F_{1,30}$ = 1.96, P = 0.17).

We found some evidence that dispersal can be increased with low severity fires. Consistent with the cover and seedling emergence data, dispersal plots that experienced the highest-severity fires had fewer postfire *B. sylvaticum* germinants, whereas the plots with the lowest-severity fires showed the highest amounts of germinants. Whether or not this is an indication that dispersal was lower in the high-severity fire areas is a bit tricky to assertain. We measured dispersal only indirectly, by tracking infestation over time

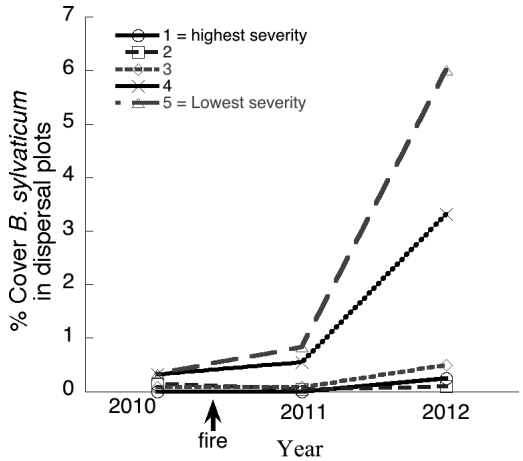

Figure 4. Dispersal. Norms of reaction graph showing the interactions among year, fire severity, and cover of *B. sylvaticum* in the dispersal plots. The points are averages; they are connected by lines to illustrate the lack of parallel lines, indicating a statistical interaction. There was a significant effect of fire severity on dispersal, with low-severity plots exhibiting greater levels of colonization ($F_{4,27} = 5.50$, $P = 0.002$).

in burned and unburned plots that prior to fire showed no sign of infestation. Although there were no or few plants to start with, this does not mean that there were no seeds present in the seed bank. Future studies should assess the seedbank in dispersal plots preburn to examine this possibility. It would also be good to examine other potential causes, such as seeds being carried on fire crews and equipment. Additionally, what we are observing could be due to a stimulation effect of fire on the seedbank. More research in this area is needed before we can say anything definitively.

We found that low-severity fires, which are typical of spring prescribed burns in the Oregon Cascades, tend to increase the cover of *B. sylvaticum* where it is already present, as well as its potential dispersal to areas where it has not previously occurred. We also found that areas with high cover of *B. sylvaticum* experience lower-severity fires. Thus a feedback loop can be initiated with low-severity fires such that the grass increases as a result, thereby decreasing severity and increasing cover; this system could potentially fit the grass–fire cycle.

Conversely, we found that high-severity burns decrease *B. sylvaticum* cover, and fire treatment in general decreases seedling emergence. It is clear that fire is not a simple tool. There is at least as much potential for fire to increase invasions of this grass as to contain it. That said, when fires do occur or are prescribed, follow-up with additional control measures in the less-severely burned zones could help to contain this invasion.

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