Project Title: Management of Fuel Loading in the Shrub-Steppe: Responses Six, Seven, and Eight Years After Treatments

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Principal Investigators:

Dr. Steven O. Link, President, Native Plant Landscaping and Restoration LLC, 4604 E. Robin Court, West Richland, WA 99353; Phone: 509/948-0054; e-mail: stevenlink@me.com

Mr. Randy Hill, Deputy Project Leader, Ridgefield NWR Complex, Ridgefield, Pierce, Franz Lake and Steigerwald Lake National Wildlife Refuges. 28908 NW Main Ave, PO Box 457, Ridgefield WA 98642; Phone: 360/887-3883/4106; fax: 360-887-4109; e-mail: randy_hill@fws.gov

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I. Abstract

Our objective was to determine if our strategy to reduce *Bromus tectorum* cover and thus fire risk is sustainable after implementation. Our primary task was to test the hypothesis that the bunchgrass, *Elymus wawawaiensis*, established in 2003 will show an increasing degree of *B. tectorum* control 6, 7, and 8 years after seeding. Six years after plots that were burned, treated with imazapic (Plateau) herbicide at a rate of 4 or 8 oz/acre, and then drill-seed with *E. wawawaiensis* had significantly ($p = 0.0016$) less *B. tectorum* cover ($14.4 \pm 4.2\%$) than plots that were only burned (37.9 ± 3.7 %). In the eighth year, the effect remained, with significantly ($p = 0.002$) less *B. tectorum* cover ($21.2 \pm 4.1\%$) in treated plots than in plots that were only burned ($46.9 \pm 4.4\%$). There was no significant change in *B. tectorum* cover in the treated plots 6, 7, and 8 years after seeding, thus the hypothesis is false. We tested the hypothesis that native species cover, native species richness, and cover of aliens in plots that were burned, treated with imazapic, but not drill-seeded will not be different from plots that were only burned. There were no significant ($p > 0.05$) differences for these responses between treated plots and burned controls 6, 7, or 8 years after treatment application. We found a strong reduction in *B. tectorum* cover with increasing density of *E. wawawaiensis.* We observed a significant effect of *E. wawawaiensis* volume on *B. tectorum* cover and distance from the bunch. *Elymus wawawaiensis* apparently interferes with the ability of *B. tectorum* to establish up to 1 m away from the centers of large plants. We also observed a significant increase in % senescing plants during the observation period, which, along with large patches of apparently senescing plants in an adjacent field suggest that establishing *E. wawawaiensis* may not be a sustainable strategy for long-term control of *B. tectorum* in the study area.

II. Background and Purpose

Land management in the western United States is complicated by fires associated with invasive species such as *Bromus tectorum* (Whisenant 1990). Establishing cool season perennial bunchgrasses has been shown to control *B. tectorum* (Klomp and Hull 1972; Thompson et al. 2006) when seeded in the spring following fall tillage (Whitson and Koch 1998). It is possible to reduce cover of *B. tectorum* and thus fire risk (Link et al. 2006a) by instituting prescribed fire management strategies, applying herbicide, and drill seeding native perennial grass species in the shrub-steppe without tillage (Link et al. 2005a). Link et al. (2005a) found, at the time, that the best strategy was to conduct a prescribed burn in the fall, apply imazapic herbicide at a rate of 4 or 8 oz/acre, and then drill-seed *Elymus wawawaiensis* (Snake River wheatgrass). Land managers at the Columbia, McNary, and Umatilla National Wildlife Refuges are now applying imazapic at levels similar to those determined in our experiments (4 and 8 oz acre^{-1}). They wish to know how long the effect of imazapic will remain and if drill-seeded grasses will be sufficient to reduce *B. tectorum* cover. Also, they are interested in determining if the increase in native species richness is sustainable after application of imazapic.

Seeding *E. wawawaiensis* at about 7 lbs acre^{-1} resulted in about 0.55 plants m^{-2} (Link et al. 2005a). We were not able to detect the effect of the newly established bunchgrasses on plant community composition or cover in 2004. It takes a longer time to recognize the competitive effect of bunchgrasses on *B. tectorum* cover. To obtain better value from the experiment started in 2002 (Link et al. 2005a) it was necessary to document responses to treatments for a longer period of time. This is relevant to the Task Statement in the AFP, "Re-measurement of past JFSP study sites or experimental plots".

The competitive effect of bunchgrasses on *B. tectorum* cover was observed in an adjacent field from our study area. After 18 years (since 1986), drilled *E. wawawaiensis* bunchgrasses (2.77 plants m^{-2}) reduced *B. tectorum* cover to 2.8% (see figure 1) compared with about 25% in adjacent untreated fields. It should be noted that *B. tectorum* cover is highly variable year-to-year ranging from about 52% in 2003 to about 25% in 2004 in untreated plots (Link et al. 2005a).

Figure 1. Eighteen years (since 1986) after drill seeding the large bunchgrass, *E. wawawaiensis,* results in near elimination of *B. tectorum*, an increase in bare soil and soil cryptogam cover, and a reduction of fire risk at the Columbia National Wildlife Refuge.

While we know *B. tectorum* cover was greatly reduced after 18 years, we do not know how long it took to achieve this level of control. There have been few attempts to address this question. Seven years after seeding a number of perennial bunchgrasses and rhizomatous grass species, effects on *B. tectorum* biomass were variable (Robertson et al. 1966). *Agropyron inerme* reduced *B. tectorum* biomass by 40% while *Elymus elymoides* reduced *B. tectorum* biomass by 88% (Robertson et al. 1966). Drill-seeding a mixture of native species including *E. wawawaiensis* resulted in significant reductions in annual weeds including *B. tectorum* three years after seeding (Thompson et al. 2006). There is little information on how long it takes for *E. wawawaiensis* alone to dominate a site (Monsen et al. 2004a). Monson et al. (2004a) also notes that there are many introduced perennial grasses that can be used to control *B. tectorum.* It will be important to land managers to know how long it will likely take to achieve a significant reduction in *B. tectorum* cover and fire risk through planting competitive bunchgrasses such as *E. wawawaiensis*. Our primary task was to test the hypothesis that bunchgrasses established in 2003 will show an increasing degree of *B. tectorum* control 6, 7, and 8 years after seeding. If there is no increasing control of *B. tectorum* over the three years of observations then managers can suggest that a higher rate of seeding is needed to establish greater control. If the current experiment shows increasing and significant control then managers can be assured that fire risk will be reduced in a predictable amount of time.

Another result of our application of imazapic at 8 oz acre^{-1} was a 58% increase in native species richness over controls and a decrease in cover of alien species two years after application (Link et al. 2005a). Little is known on how long communities treated with imazapic will continue to show effects. Beran et al. (1999) showed increases in native wildflower density two years after imazapic (Plateau) application. Similarly, Beran et al. (2000) showed increases in *Andropogon gerardii* yields two years after applying imazapic. Long-term effects of herbicides have been noted 11 years after application (Miller et al. 1999). We tested the hypothesis that native species richness and cover in burned imazapic plots without drill-seeded *E. wawawaiensis* will not be different from burned control plots 6, 7, and 8 years after treatment application. We also tested the hypothesis that alien species cover in burned imazapic plots without drill-seeded *E. wawawaiensis* will not be different from burned control plots 6, 7, and 8 years after treatment application. We asked these questions to see if increases in native species and reductions in alien species cover are maintained up to 8 years after imazapic application. If this is true, then part of the hypothesized reduction in *B. tectorum* cover associated with *E. wawawaiensis* may be attributed to the effect of imazapic alone. If native species richness increases are maintained and *B. tectorum* and other invasive species cover is significantly reduced over the three-year observation period then the simple use of imazapic without drill seeding bunchgrasses may be sufficient to reduce fire risk.

Our objective was to determine if our strategy to reduce *B. tectorum* cover and thus fire risk is sustainable after implementation. We monitored species composition and cover in 6 plots without imazapic application, but with and without drill seeded bunchgrasses and in 6 plots where imazapic had been applied at 4 and 8 oz acre^{-1} with and without drill seeded bunchgrasses. All plots were burned in October 2002. All plots are split-plots. We also monitored the survivorship, flowering, condition, and size of established bunchgrasses. We documented new *E. wawawaiensis* arising from self-seeding. We also describe the relationship between *E. wawawaiensis* canopy volume and interference distance with *B. tectorum.* This was done to characterize the spatial extent of *E. wawawaiensis* and its abilities to reduce seed production (Weiner 1982) of *B. tectorum.*

III. Study Description and Location

Study Area

The Columbia National Wildlife Refuge is located in the Columbia Basin of eastern Washington. It includes more than 23,000 acres (9,308 ha) north and west of Othello in Grant and Adams counties, and lies in the rain shadow of the Cascade Mountains. Precipitation averages 8" (203 mm) per year, with most rain falling between October and April. Snowfall is quite variable with winter high temperatures usually near freezing. Lightning during the summer is a frequent cause of wildfire.

Cover was dominated by *Artemisia tridentata* (big sage), *Poa secunda* (Sandberg's bluegrass), and *Pseudoroegneria spicata* (bluebunch wheatgrass) when cattle and sheep were introduced in the 1800's. The area was severely overgrazed and soon dominated by *B. tectorum*. The Columbia Basin Irrigation Project brought water to the area in the 1950's, and the refuge area was set aside due to rockiness, shallow soils, and depression areas that filled with water from a rising water table and seepage from reservoirs and canals. Grazing was halted more than 20 years ago and fire is the most prominent disturbance to upland areas. In untreated areas *B. tectorum* remains the dominant cover, with variable amounts of annual and perennial forbs. All areas have *P. secunda* and scattered *P. spicata*.

Experimental Design and Treatment Application

Twelve plots (8.2 m x 33 m) were established in spring 2002 (Link et al. 2005a). The study area was 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. Treatments were then randomly applied to the plots. Imazapic was applied on Nov. 14, 2002 as a pre-emergent at five concentrations including 4 and 8 oz acre^{-1} or 0.28 and 0.56 kg ha^{-1} with a boom sprayer (Spider Sprayer, West Texas Lee Company). Imazapic were applied using water at a rate of 281 kg ha^{-1}. There are three replicates for each level of imazapic. Each plot was then split and *E. wawawaiensis* was drill seeded in half the plot. *Elymus wawawaiensis* (Snake River wheatgrass, Secar cultivar) was drill seeded at a rate of about 7.9 kg ha^{-1} (about 215 PLS m^{-2}) on Feb. 19, 2003. Drill rows were 0.3 m apart. The old name for the Secar cultivar was *P. spicata* (Carlson and Barkworth 1997). Early observations were taken in 2002, 2003, and 2004 (Link et al. 2005a).

Measurements

Observations and measurements were taken in 2008, 2009, and 2010. Species richness was determined in each plot by identifying all vascular plant species. This was done by inspection in the spring and summer.

Cover was determined using a tape (Bonham 1989; Elmore et al. 2003; Link et al. 2005b) and identifying the first observed (tallest) cover type at each 0.25 m hash mark on the tape. All drill-seeded grasses and their progeny were counted, flowering status (in flower or not) recorded, senescence status (appearance of gray or dying leaves and tillers; see figure 4) recorded, and measured for greatest height in each plot. A meter stick was used to measure height.

The influence of individual bunch size on *B. tectorum* was determined at 15 *E. wawawaiensis* plants across all sizes. Cover associated with individual bunches was determined every 5 cm away from the bunch center. This assessment was done along a randomly placed tape across the center of a bunch. Readings were taken away from both sides of the center of the bunch until there was no longer any apparent interference from the bunchgrass on *B. tectorum*. A second layer was identified under the canopy of *E. wawawaiensis.* Cover in control plots where there were no *E. wawawaiensis* plants was also assessed as a control. Cover every 5 cm was taken along each of six transects. Each transect was 1.5 m long yielding 30 observations.

The greatest distance until bunchgrass interference was lost was defined as three to five repeated *B. tectorum* observations along the tape. This distance was determined on both sides of the bunch and then averaged for an interference distance for each plant. The average interference distance was then related to bunch volume. Bunch volume was determined by measuring the greatest height as done for all other bunchgrass observations, the diameter at the top of the canopy along the tape and the basal diameter along the tape.

Parameter Estimation

The relation between % *B. tectorum* cover and % *E. wawawaiensis* cover is:

$$Y = (cover_{max} - cover_{min})e^{(-K*x)} + cover_{min}, \tag{1}$$

where Y is % *B. tectorum* cover, $cover_{max}$ is estimated cover when % *E. wawawaiensis* cover = 0, $cover_{min}$ is estimated cover for high % *E. wawawaiensis* cover, K is rate constant, and x is % *E. wawawaiensis* cover.

The relationship between percent senescent plants and years after seeding is:

$$\%\ \text{senescent plants} = \alpha e^{kt}, \tag{2}$$

where α estimates initial % senescent plants, k is the rate constant, and t is time (years).

Canopy volume was determined by assuming the bunchgrass is a conic section (frustum) with the small cut section at the base.

Volume (V, m^3) is computed as:

$$V = \pi h(\alpha^2 + \alpha\beta + \beta^2)/12, \tag{3}$$

where h is the greatest canopy height (m), β is the basal diameter (m), and α is the diameter at the top of the canopy (m).

The relationship between volume (V, m^3) and interference distance (d, m) is:

$$d = d_{min} + (d_{max} - d_{min})(1 - e^{(-kV)}), \tag{4}$$

where d_{min} is the estimated minimal interference distance (m) when V is zero, d_{max} is the estimated maximal interference distance (m) for large plants, and k is a constant (m^{-3}).

Data analysis

Each plot (whole plot) was randomly assigned an herbicide treatment. The split-plot is an experimental unit for drill seeding. All observations within each split-plot were averaged or summed for analysis. Statistical analyses were done using JMP version 9, software (SAS Institute 2010). Percent cover data were transformed by:

$$arcSin\sqrt{\frac{\%\,cover}{100}}, \tag{5}$$

before statistical analysis (Steele and Torrie 1960). The whole plots are nested within herbicide treatment. The subplot factor is drill seeding. Effects were assessed using a least squares means differences Tukey HSD test. Species richness count data are likely to have a Poisson distribution, thus were transformed with the natural log for analysis (MacNally and Fleishman 2004). Untransformed species richness count and percent cover data are presented to facilitate interpretation. Statistical significance is set at the $\alpha = 0.05$ level.

III. Key Findings

Native plant species richness as affected by imazapic

Native species richness in burned imazapic treated plots (without drill-seeded bunchgrasses) was not significantly different than in burned control plots in 2008, 2009, or 2010 (Table 1). This is in contrast to the significant increase in native species richness with imazapic treatment in the second year after application (2004).

Table 1. Native species richness in burned imazapic treated and burned control plots (n = 6). Observations in 2002 were made before imazapic treatments were applied to the study plots. Observations after 2002 were after the fire and imazapic treatment application.

Year	Control (± 1 sem)	Imazapic (± 1 sem)	p
2002	8.2 ± 0.54	9.0 ± 1.1	0.51
2003	12 ± 0.43	13 ± 0.87	0.20

2004	11 ± 0.76	17 ± 0.91	0.0009
2008	11 ± 1.0	13 ± 0.63	0.21
2009	12 ± 0.83	13 ± 0.98	0.33
2010	10 ± 1.4	12 ± 1.1	0.082

We observed no effect of imazapic application on native species richness the year after application, a significant increase in the second year after application, returning to no effect during the three years of the current study. Imazapic application under the conditions of our study apparently released the native flora from competition allowing more species to establish, but only in the second growing season after application. There is little value in applying imazapic alone or after a prescribed fire to increase native species under the conditions of these studies. It is possible the seed bank was increased which may improve opportunities for the native flora to re-establish under future conditions. This has not been investigated.

Bromus tectorum cover and control, native species, and alien species cover

Six, 7, and 8 (2010) years after plots were burned, treated with imazapic at a rate of 4 or 8 oz/acre, and then drill-seeded with *E. wawawaiensis* there was significantly less *B. tectorum* cover than burned control plots (Fig. 2). Cover of *E. wawawaiensis* was 9.5 ± 2.2% in 2008, 11 ± 3.1% in 2009, and 12 ± 3.8% in 2010, which are all significantly ($p < 0.05$) greater than zero. Addition of imazapic alone after burning did not result in a significant decrease in *B. tectorum* cover compared with the burned controls in 2008, 2009, or 2010 (Fig. 2). Mean *B. tectorum* cover in the burned control plots ranged from 38 to 47% across the three years demonstrating natural yearly variation. There was no significant change over the three years in *B. tectorum* cover in the burned control ($p = 0.12$) or in the burned treated plots ($p = 0.23$).

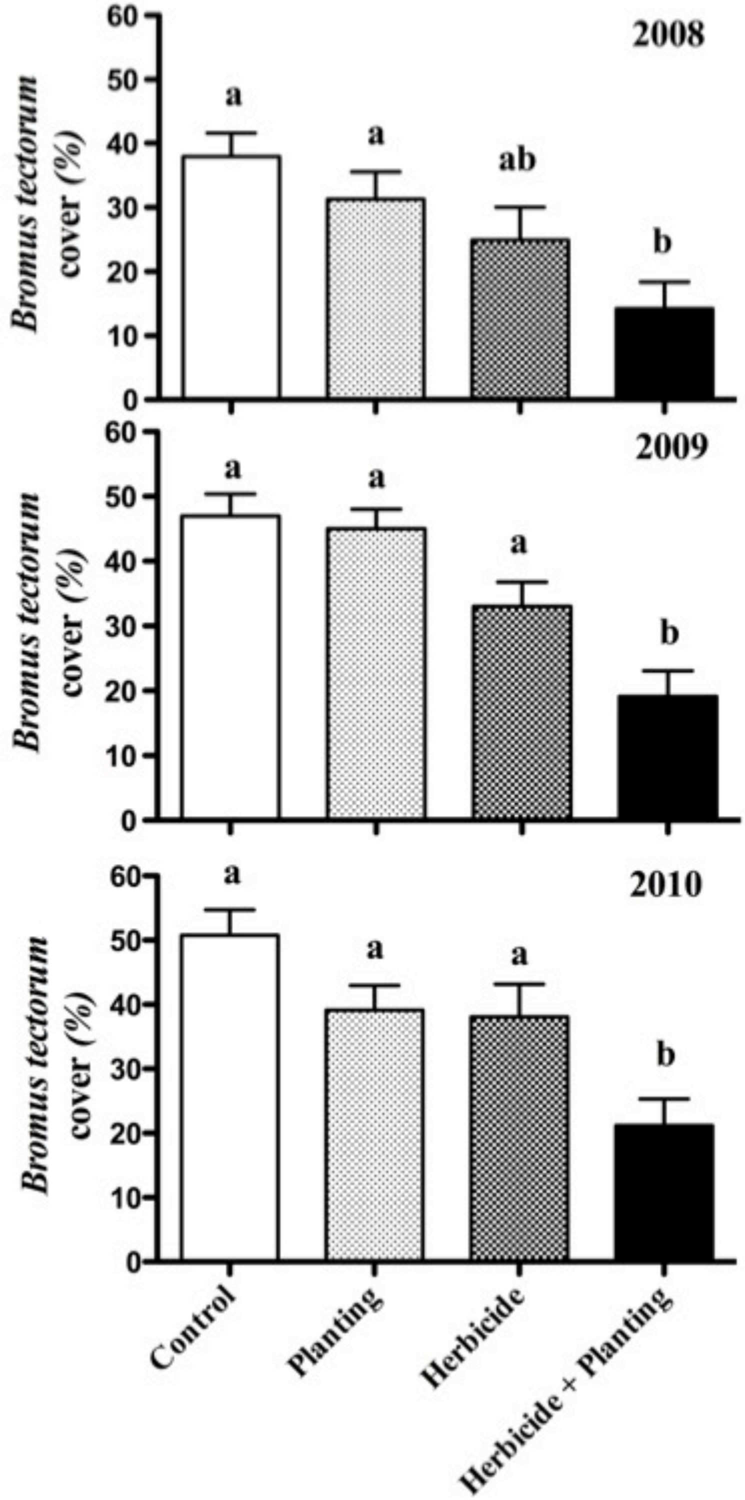

Figure 2. Effect of treatments on *B. tectorum* percent cover (mean ± 1 sem) in 2008, 2009, and 2010. Differing letters indicate significant differences.

There was no significant ($p > 0.05$) difference between burned imazapic without seeding and burned control plots on native or alien species cover in any year from 2004 through

2010. Combining burned imazapic and burned control plots revealed no change (p = 0.61) over six years in native species cover (Fig. 3) while there was a significant (p = 0.0005) increase in alien species cover over the six years (Fig. 4).

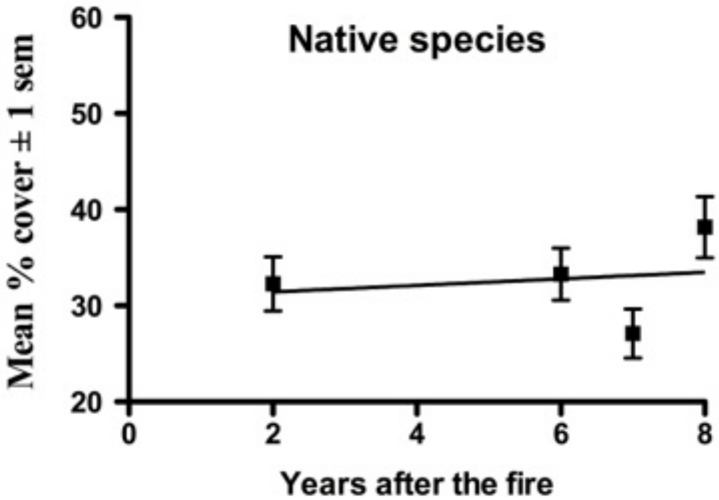

Figure 3. Native species cover from 2004 through 2010. The slope of the linear regression over time was not significantly (p = 0.061) different from zero.

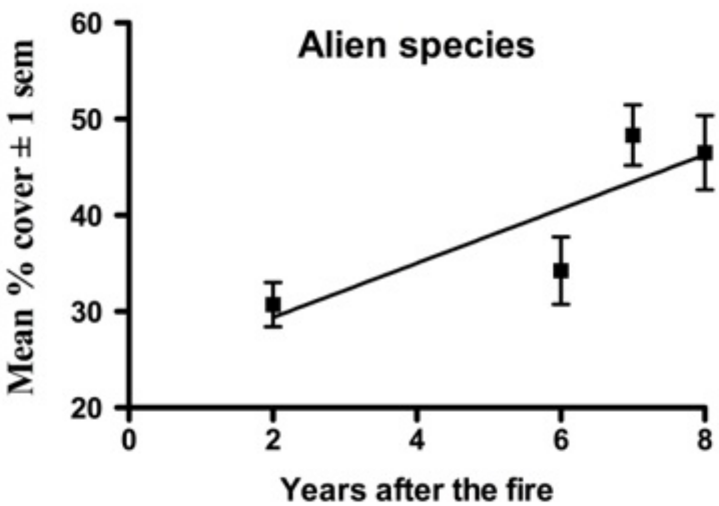

Figure 4. Alien species cover from 2004 through 2010. The slope of the linear regression over time was significantly (p = 0.0005) greater than zero.

We observed no change in native species cover and a significant increase in alien species cover in unseeded plots 8 years after the fire. The significant increase in alien species cover was predominately that of *B. tectorum.*

When % *B. tectorum* cover was related to % *E. wawawaiensis* cover using all 12 plots (with and without imazapic, but all seeded) a significant ($p = 0.016$, $r^2 = 0.46$) and exponentially decreasing relationship (Eq. 1) was found ($cover_{max} = 32 \pm 4.9\%$, $cover_{min} = 10 \pm 10\%$, $k = 0.22 \pm 0.31$) in 2008 (Fig. 5). A similar (Fig. 5) and significant ($p = 0.0006$, $r^2 = 0.71$) response was found ($cover_{max} = 45 \pm 4.0\%$, $cover_{min} = 7.7 \pm 13\%$, $k = 0.12 \pm 0.09$) in 2009 and in 2010 ($p = 0.0027$, $r^2 = 0.61$, $cover_{max} = 41 \pm 4.3\%$, $cover_{min} = 9.4 \pm 12\%$, $k = 0.10 \pm 0.084$).

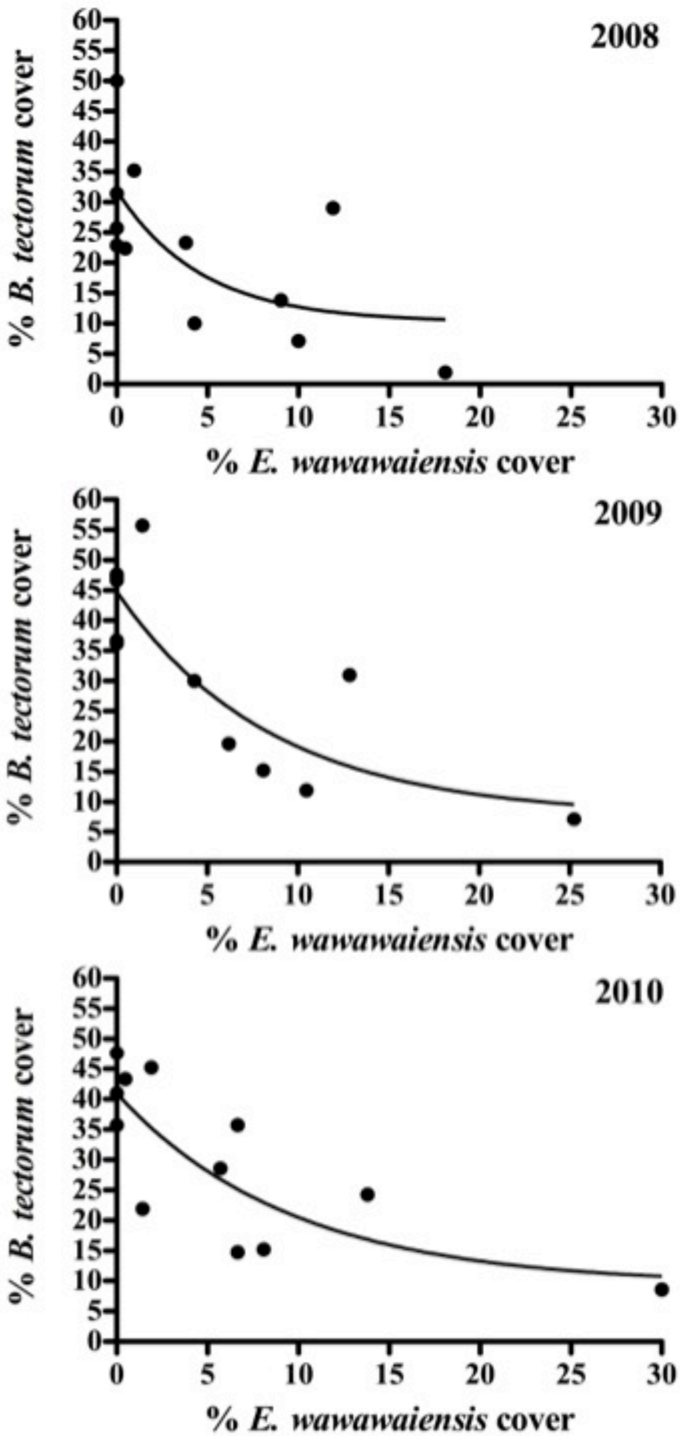

Figure 5. The relationship between % *B. tectorum* cover and % *E. wawawaiensis* cover 6, 7, and 8 years after seeding.

We found a long-term and significant reduction of *B. tectorum* 8 years after treatment application and drill-seeding *E. wawawaiensis*. *Bromus tectorum* cover was reduced from 47% in burned control plots to 21% in plots with *E. wawawaiensis*. Compared with burned control plots we achieved a 55% reduction of, or 'control' of *B. tectorum*. We seeded 95 days after application, which was, apparently, long enough to significantly reduce *B. tectorum* and allow successful establishment of *E. wawawaiensis*.

Population dynamics, flowering, condition, and size of established bunchgrasses

All seeded plots with imazapic had significant populations of *E. wawawaiensis* eight years (2010) after seeding. There was no significant ($p = 0.15$) linear increase in the population over time (Fig. 6). The population has stabilized through 8 years after planting. Changes from year 7 to year 8 were two plots increasing a total of 12 plants and four plots decreasing a total of 21 plants. The number of recognized seedlings or very small plants was low: one in 2008, four in 2009, and three in 2010.

Seeded plots that were not treated with imazapic had very low populations. Two years after planting there was an average of 1.3 ± 1.1 plants per plot, but this mean is not significantly different from zero. In years 6, 7, and 8 there were significant, yet very small populations of 2.0 ± 0.6, 4 ± 0.6, and 4.7 ± 0.7 plants per plot respectively. There was a significant ($p = 0.0065$) linear increase in these populations over the time period.

Combining counts of all *E. wawawaiensis* plants in all 12 plots revealed 694 plants in 2004, 946 plants in 2008, 1022 plants in 2009, and 1017 plants in 2010. There is continuing recruitment with no recruitment outside the drill row disturbance. Continuing recruitment suggests delayed germination or that the drill row disturbance favors incorporation and germination of new seed.

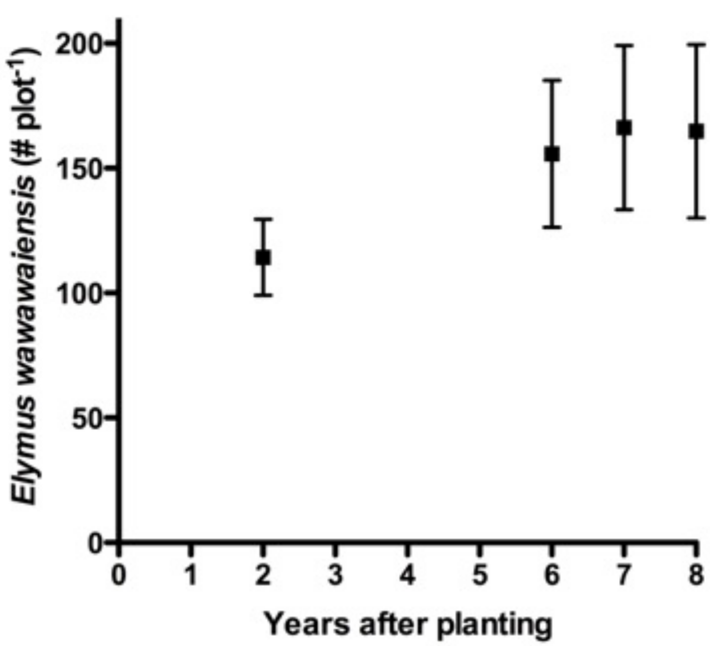

Figure 6. The mean number of *E. wawawaiensis* bunchgrasses per plot. The error bars are one standard error of the mean (n = 6).

The numerical increases in some of the plots since two years after planting (Fig. 6) suggest that either the original seed had germination delays or that there was recruitment of new individuals from new seed. There were no individual bunches found outside the drill-rows suggesting delayed germination or that the drill row disturbance strongly favors incorporation and germination of new seed. While it was possible to recognize a few very small plants or seedlings in each observation year it is also possible that counting errors can account for some of the variation in populations in the three years. There has been very little research on the effect of the disturbance caused by seed drilling on subsequent germination.

While the population apparently has stabilized there is an increasing number of apparently senescent plants (Figs. 7 and 8).

Figure 7. *Elymus wawawaiensis* 8 years after seeding showing senescence (gray tillers) with remaining green tillers mostly near the edge.

The relationship (Eq. 1) between % senescent plants in the imazapic plus seeded plots and years after planting (Fig. 8) was significant ($p = 0.0082$, $r^2 = 0.28$, $\alpha = 0.018 \pm 0.07$, $k = 0.86 \pm 0.5$). The high variability eight years after planting is because one plot had 60% of plants showing signs of senescence. Changes from year 7 to year 8 were two plots with a reduction and four plots with an increase in percent senescence.

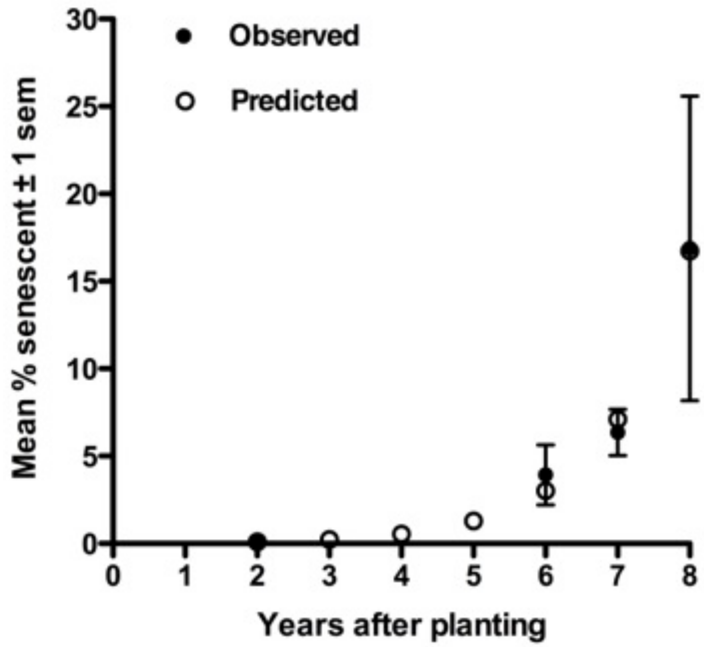

Figure 8. Mean and predicted % senescent plants (± 1 sem) in the imazapic seeded plots.

Apparently senescent populations were observed in nearby areas that had been seeded in 1986 (Fig. 9). The populations that were in a senescent condition were patchy in distribution throughout the roughly 36 ha area.

Figure 9. Gray colored senescing *E. wawawaiensis* seeded in 1986 near the study area on September 24,2009.

The percent of plants in flower has increased linearly and significantly ($p = 0.0045$) from two years after planting (42 ± 3.6) to 8 years after planting (60 ± 4.3) for the 6 imazapic and seeded plots. The number of flowering culms on individual bunches, while not counted, varied from one to numerous allowing for production of numerous seed.

The percent of *E. wawawaiensis* in flower increased from two to 8 years after seeding. It is likely that plants in the current experiment gradually increased flowering as they became larger.

The greatest height of bunches was compared across years for those plants in flower and those not in flower in the 6 imazapic seeded plots. The greatest height of plants in flower was not significantly ($p = 0.29$) different among years and ranged from 68 ± 1.6 cm two years after planting to 73 ± 2.4 cm 8 years after planting. The greatest height of plants not in flower increased linearly and significantly ($p = 0.0004$) from two years after planting (37 ± 0.56) to eight years after planting (43 ± 1.2).

Elymus wawawaiensis size and interference with B. tectorum

There was a significant ($p < 0.0001$) effect of bunchgrass size (canopy volume) on the interference distance with *B. tectorum* (Eq. 3, Fig. 10). As bunchgrass canopy volume increases, the distance away from the center of the bunch with low *B. tectorum* cover increases. The maximal estimated interference distance (d_{max}, Eq. 3), is the apparent negative effect of very large bunchgrasses on *B. tectorum* establishment up to 1.2 ± 0.33 m away from the bunchgrass center. Estimated d_{min} is 0.086 ± 0.17 which is not significantly different from zero. The estimated curvature constant k is 4.0 ± 3.0.

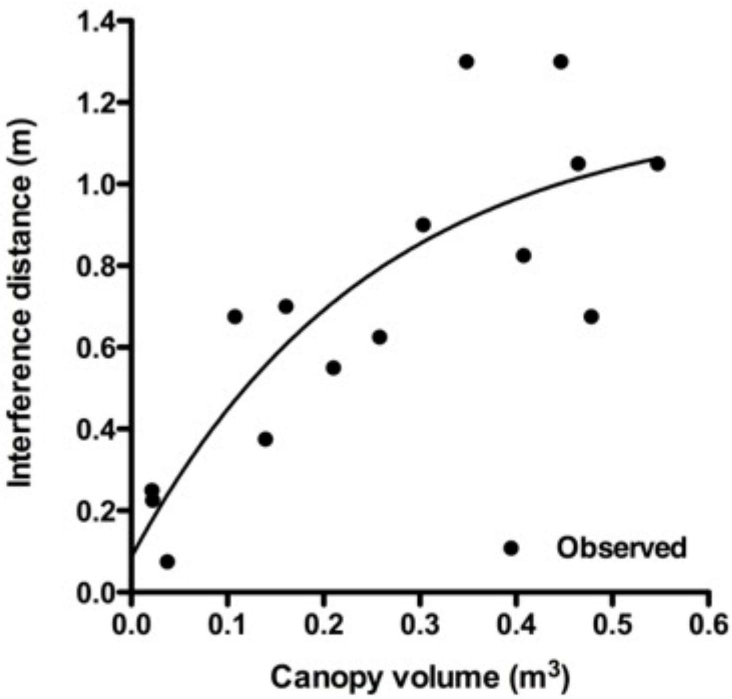

Figure 10. The relationship between *E. wawawaiensis* canopy volume and the interference distance with *B. tectorum*.

The effect of *E. wawawaiensis* interference on *B. tectorum* was also assessed based on *B. tectorum* cover. When there is no *E. wawawaiensis*, cover of *B. tectorum* was 82 ± 2.8 % (n = 6). When *E. wawawaiensis* exerts significant control within the defined interference distance from bunch centers, *B. tectorum* cover was 11 ± 2.5 % (n = 15). Beyond the determined effect of *E. wawawaiensis*, *B. tectorum* cover was 82 ± 3.4 % (n = 15), which is the same as areas without *E. wawawaiensis*. Thus, we have adequately defined the spatial extent of *E. wawawaiensis* interference on *B. tectorum* for this study site.

We observed a significant effect of *E. wawawaiensis* volume on *B. tectorum* cover and distance from the bunch. *Elymus wawawaiensis* apparently interferes with the ability of *B. tectorum* to establish or produce seed.

Density of E. wawawaiensis, B. tectorum cover, and fire risk

Bromus tectorum cover decreases with increasing density of *E. wawawaiensis* (Fig. 11). The highest *B. tectorum* cover occurred in control plots without *E. wawawaiensis.* The natural variability in cover in the control plots was reflected in the variability over the 6 to 8 years of *B. tectorum* cover in plots with *E. wawawaiensis*. The highest density *E. wawawaiensis* and lowest cover of *B. tectorum* were observed 18 years after seeding in the adjacent field that had been planted in 1986 after a fire (Link et al. 2005).

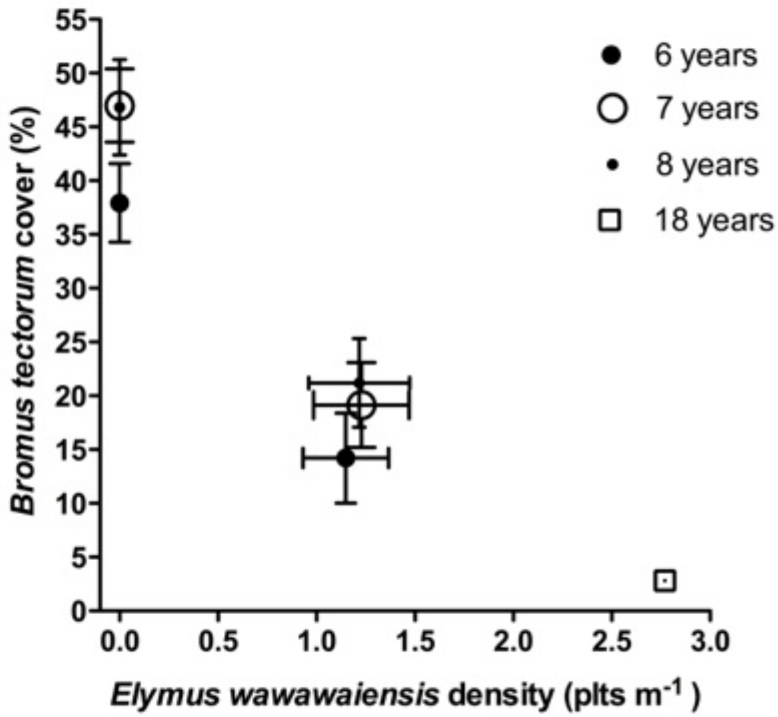

Figure 11. Relation between planted bunchgrass density and *B. tectorum* cover.

Using the relationship between fire risk and *B. tectorum* cover derived by Link et al. (2006a) and assuming the relationship is similar for differing locations it can be concluded that fire risk is between about 88 and 95% % when *B. tectorum* cover ranges from 38 to 46%. The fire risk in the adjacent field with 2.8% *B. tectorum* cover and high *E. wawawaiensis* density was determined to be 66% (Link et al. 2005). Without *E. wawawaiensis*, fire risk with *B. tectorum* cover between 14 and 21% was between 50 and 60% Link et al. (2006a). The fire risk of the plots with *E. wawawaiensis* density between 1.1 and 1.3 plants m^{-2} and *B. tectorum* cover between 14 and 21%, while not tested, may be between 60% and 70%.

V. Management Implications

We demonstrated that *B. tectorum* cover can be significantly reduced for at least 8 years after burning and applying imazapic at 4 or 8 oz acre^{-1} in the fall and then drill seeding *E. wawawaiensis* in mid-February. Burning and applying imazapic without drill seeding *E. wawawaiensis* had no effect on native and alien species cover. Burning and applying imazapic without drill seeding *E. wawawaiensis* had no effect on native species richness 6 to 8 years after treatment. Eight years after treatment application the density of drill seeded *E. wawawaiensis* was 1.2 ± 0.26 plants m^{-2}. Doubling the seeding rate may increase density enough to further reduce *B. tectorum* cover to be closer to the minimal value of 2.8% achieved with *E. wawawaiensis* density of 2.8 plants m^{-2}. It was estimated that the largest solitary *E. wawawaiensis* plants significantly interfere with *B. tectorum* to an estimated distance of about 1.2 m from the bunch center. The *E. wawawaiensis* populations have stabilized statistically 6 to 8 years after seeding reducing *B. tectorum* cover to about 21% compared with cover in control plots of about 47% eight years after planting. While *E. wawawaiensis* populations have stabilized statistically 8 years after seeding, about 17% of the population has apparently become senescent. The pattern of apparently senescent populations was patchy across the landscape and was also observed in populations seeded in 1986. This observation suggests that *E. wawawaiensis* populations are not likely to sustain themselves at the Columbia National Wildlife Refuge. It is suggested that seed of *P. spicata* (bluebunch wheatgrass) that naturally exists in the study area be considered for long-term and sustainable control of *B. tectorum* at the Columbia National Wildlife Refuge.

VI. Relationship to Other Recent Findings and Ongoing Work on This Topic

Native plant species richness as affected by imazapic

We observed no effect of imazapic application on native species richness the year after application, a significant increase in the second year after application, returning to no effect during the three years of the current study. Imazapic application under the conditions of our study apparently released the native flora from competition allowing more species to establish, but only in the second growing season after application. This is in contrast to the observation of decreased species richness one year after application with a return to control levels in the second year (Owen et al. 2011). Beran et al. (1999a) observed little effect of imazapic application at 70 g ai ha^{-1} on seeded wildflower establishment 14 months after treatment. Beran et al. (1999b) found imazapic application at 70 g ai ha^{-1} before seeding legumes reduced weed interference and improved establishment of some of the legume species 14 months after treatment. Elseroad and Rudd (2011) observed little effect of applying imazapic at 70 g ai ha^{-1} on native flora four years after treatment application and conclude that reducing *B. tectorum* alone was not an effective strategy for increasing native perennials. While we conducted a prescribed burn and applied higher levels of imazapic (280 and 560 g ai ha^{-1}) observing significant increases in native species richness in the second year after application, the effect disappeared six years after application and thereafter. There is little value in applying imazapic alone or after a prescribed fire to increase native species under the conditions of these studies. It is possible the seed bank was increased which may improve opportunities for the native flora to re-establish under future conditions. This has not been investigated.

Bromus tectorum cover and control, native species, and alien species cover

We found long-term and significant reduction of *B. tectorum* 8 years after treatment application and drill-seeding *E. wawawaiensis*. *Bromus tectorum* cover was reduced from 47% in burned control plots to 21% in plots with *E. wawawaiensis.* Compared with burned control plots we achieved a 55% reduction of, or 'control' of *B. tectorum.* Our results differ from that of Morris et al. (2009) who observed decreasing *B. tectorum* cover for only one of five seeded perennial species at each of two sites. The only perennial to decrease *B. tectorum* cover by the third year was *Agropyron fragile* ('Vavilov'). We observed significant reductions 6, 7, and 8 years after treatment application and it is possible that levels of control in Morris et al. (2009) could increase with increasing plant size or number. Whitson and Koch (1998) observed reductions of *B. tectorum* ranging from 100% to 32% for five species of cool-season perennial grasses three years after seeding. Shinn and Thill (2004) found excessive injury to *E. wawawaiensis* seedlings when seeded one day after imazapic application at levels that control *B. tectorum.* We seeded 95 days after application, which was, apparently, long enough to significantly reduce *B. tectorum* and allow successful establishment of *E. wawawaiensis*.

We observed no change in native species cover and a significant increase in alien species cover in unseeded plots 8 years after the fire. The significant increase in alien species cover was predominately that of *B. tectorum.* This result is similar to that of others (Hosten and West 1994; Ott et al. 2001; Young and Evans 1978) who observed significant increases in *B. tectorum* after a fire.

Population dynamics, flowering, condition, and size of established bunchgrasses

There has been very little population dynamics research on the effect of the disturbance caused by seed drilling on subsequent germination of *E. wawawaiensis.*

We observed increasing apparent senescence of *E. wawawaiensis* with time since seeding in 2003 and also noted significant and large-scale apparent senescence of *E. wawawaiensis* seeded in 1986. Planting density of the grasses could be a factor in the senescence observations because Humphrey and Pyke (1998) found a strong and decreasing relationship between tiller number, biomass, and probability of flowering and increasing density. The density relationship in Humphrey and Pyke (1998) changed rapidly at very low-density values similar to those in our study. Large decreases in response occurred at much higher densities (Humphrey and Pyke 1998) than in our study, thus it is not likely that senescence in our study was associated with competition for resources. Local extinctions of perennial grasses have been associated with extended drought especially in semi-arid regions (O'Connor T.G. 1991). *Elymus wawawaiensis* is expected to survive dry conditions with yearly precipitation down to 203 mm (Ogle et al 2008). The long-term precipitation average for nearby Othello, Washington is 208 mm (Western Regional Climate Center, 2006; http://www.wrcc.dri.edu/), which is sufficient for this grass. *Elymus wawawaiensis* has established and sustained itself in the drier (average yearly precipitation of 177 mm) conditions of the Hanford Site for 15 years (Ward et al. 2010) implying that *E. wawawaiensis* can tolerate yearly precipitation less than 203 mm. Thus, low precipitation is not a likely cause for the senescence observed in the current study. It is possible that *E. wawawaiensis* planted in 1986 is reaching the end of it's lifespan because maximum plant longevity of bunchgrasses is considered not to exceed 30 years (Briske 1991). This would not account for the increasing senescence observed for plants seeded in 2003. These bunchgrasses did not exhibit classic hollowing out of the bunchgrass center (Briske 1991) associated with old plants (Fig. 4). Other possible reasons for the patchy appearance of senescent plants may be associated with patchy distribution of causal factors such as perhaps shallow soils or pathogens (Nelson 2004). The increasing observations of senescing plants at the study site suggests that the long-term sustainability of *E. wawawaiensis* can not be generalized as in Monsen et al. (2004b).

While this discussion of senescence of *E. wawawaiensis* was based on observations taken on September 24 2009 and earlier in 2010, casual observations taken on May 31, 2013 revealed little obvious patchy senescence in *E. wawawaiensis* populations at the site. More evenly distributed senescent plants were still observed in the community. The appearance of senescence in the patchy areas (Fig. 9) may only be an indication of stress in the patchy area, but not strong enough to kill the plants. This condition may have been more obvious late in the season when the picture was taken than on May 31 when plants were green.

The percent of *E. wawawaiensis* in flower increased from two to 8 years after seeding. This observation is consistent with that of Humphrey and Pyke (1998) who noted that *E. wawawaiensis* had slow growth compared with clonal *Elymus lanceolatus ssp. lanceolatus*, investing in ramet production and delaying flowering until later years. It is likely that the plants in the current experiment gradually increased flowering as they became larger.

Elymus wawawaiensis size and interference with B. tectorum

We observed a significant effect of *E. wawawaiensis* volume on *B. tectorum* cover and distance from the bunch. *Elymus wawawaiensis* apparently interferes with the ability of *B. tectorum* to establish. Using canopy volume to define interference between species has been noted as an improvement over using density to characterize the effects of interference (Bussler et al. 1995). Possible mechanisms for the interference of *E. wawawaiensis* on *B. tectorum* are associated with the different life history traits of the two species. An established perennial is more likely to have primary access to resources having a negative effect on germinating annual seeds (Corbin and D'Antonio 2004).

VII. Future Work Needed

The long-term sustainability of bunchgrass plants needs to be examined. If they are not sustainable then the *B. tectorum* populations likely will return. We observed very little if any naturally reseeded *E. wawawaiensis* outside of the drill rows. We also observed a significant increase in percent apparent senescing plants during the observation period, which, along with large patches of apparently senescing plants in an adjacent field suggest that establishing *E. wawawaiensis* may not be a sustainable strategy for long-term control of *B. tectorum* in the study area. This statement is based on observations taken before 2013. A more sustainable strategy may be to increase populations of *Pseudoroegneria spicata* that naturally exists at the site. The value of using locally sourced seeds needs to be compared with how well plants originating from great distances from the area of concern sustain themselves.

There is little value in applying imazapic alone or after a prescribed fire to increase native species under the conditions of these studies. It is possible the seed bank was increased which may improve opportunities for the native flora to re-establish under future conditions. This has not been investigated.

We seeded 95 days after imazapic application, which was, apparently, long enough to significantly reduce *B. tectorum* and allow successful establishment of *E. wawawaiensis*. The timing of application of imazapic and timing of subsequent seeding of perennial bunchgrasses needs to be examined.

We observed a significant effect of *E. wawawaiensis* size on *B. tectorum* cover and distance from the bunch. *Elymus wawawaiensis* apparently interferes with the ability of *B. tectorum* to establish. Questions on this topic can examine the mechanism of such interference. Is the mechanism related to competition for water and nutrients or perhaps there is allelopathic effect of the bunchgrass on *B. tectorum*. What is the interference distance of other bunchgrass species? Answers to these questions can assist with determining how densely bunchgrasses need to be re-established.

Another question is how much is fire risk reduced when *B. tectorum* dominated communities are restored with perennial grasses. The lowest determined risk (Link et al. 2006a) was 46% with 12% *B. tectorum* cover in a natural *Poa secunda* dominated community. Fire risk in communities seeded with a variety of perennial bunchgrasses and with *B. tectorum* cover from 2.6 to 18.5% ranged from 66 to 78% (Link et al. 2005). It may be useful to determine fire risk in a range of natural perennial bunchgrass communities and compare this with fire risk in a range of reseeded perennial bunchgrass communities. It is likely that fire risk will be the least with small bunchgrasses such as *P. secunda* and greatest with dense large bunchgrasses. It may be possible to adjust seeding or planting rates and species (*P. secunda* and a large bunchgrass) to result in optimal large bunchgrass density (1289 plants acre^{-1} based on an interference distance of 1 m) with a natural density of *P. secunda* to reduce fire risk below that achieved with high density seeding rates of large bunchgrasses.

VIII. Deliverables Cross-Walk

Proposed	**Delivered**	**Status**
Newsletter articles	See workshops and conferences, communication elements	Not done
Annual reports	(1) Link, S. O. and R. W. Hill. 2009. Management of fuel loading in the shrub-steppe: Responses six years after treatments. (Progress Report) **http://nativeplantlandscaping.com/Files.html**	Delivered
	(2) Link, S. O. and R. W. Hill. 2010. Management of fuel loading in the shrub-steppe: Responses six and seven years after treatments.	Delivered

	(Progress Report) **http://nativeplantlandscaping.com/Files.html** (3) Link, S. O. and R. W. Hill. 2013. Management of fuel loading in the shrub-steppe: Responses six, seven and eight years after treatments. (Final Report) **http://nativeplantlandscaping.com/Files.html**	Delivered
Communication elements	Project Website **http://nativeplantlandscaping.com/Files.html**	Delivered
Workshops, conferences	(1) Link, S. O. 2013. Wildfire Prevention - Invasives as the Enemy. Given to the Washington Association of District Employees (WADE) Conference Leavenworth, WA. June 17-19, 2013, (15 audience) **http://nativeplantlandscaping.com/Files.html** (2) Link, S. O. 2013. Management of Cheatgrass Fuel Loading in the Shrub-Steppe. Given to the Northwest Fire Science Consortium, USFWS, Othello, WA. May 29, 2013 (26 audience) **http://nativeplantlandscaping.com/Files.html** (3) Link, S. O. 2013. Management of Cheatgrass Fuel Loading in the Shrub-Steppe. Webinar for JFSP http://youtu.be/QCOGO7n8O4Y Feb. 12, 2013. (~500 attendees and views) **http://nativeplantlandscaping.com/Files.html** (4) Link, S. O. 2013. Fire Ecology and Shrub-steppe Restoration. Given to WSU Extension Douglas County, Douglas County Fire Restoration Education Event. Grande Coulee, WA. February 8, 2013 (~40 audience) **http://nativeplantlandscaping.com/Files.html** (5) Link, S. O., R. W. Hill, 2011. Management of fuel loading in the shrub-steppe: Responses six and seven years after treatments. Given at the Linking Science and Management Improving Restoration Success in the Shrub Steppe Workshop, USFWS, Kennewick, WA, April 26-28, 20 attendees (6) Link, S. O., R. W. Hill, 2011. Management of fuel loading in the shrub-steppe: Responses six and seven years after treatments. Given at the 4th SER International World Conference on Ecological Restoration, Merida, Mexico August 21-25. (~40 attendees) (7) Link, S. O., R. W. Hill. 2010. Management of fuel loading in the shrub-steppe: Responses six years after treatments. Society for Ecological	Delivered Delivered Delivered Delivered Delivered Delivered Delivered

	Restoration Northwest Chapter & the Washington Chapter of The Wildlife Society Regional Conference. February 16-19, 2010. Tulalip, WA. (~30 attendees)	
	(8) Link, S. O. and R. W. Hill. 2009. Management of fuel loading in the shrub-steppe: Responses six years after treatments. AFE 4th International Congress: Fire Ecology and Management, Fire as a Global Process. Nov. 30 – Dec. 4, Savannah, GA.	Delivered
	(9) Link, S. O. 2009. Ecological Restoration in the Columbia Basin. Society of Ecological Restoration British Columbia conference: Shared Responsibility for a Sustainable Landscape, Nov. 6, 2009, Naramata, BC, (~50 attendees)	Delivered
	(10) Link, S. O. 2009. Ecological Restoration in the Columbia Basin. Given to the City of Oliver Parks Department, Oliver, BC, Nov. 5, 2009. (5 audience)	Delivered
	(11) Link, S. O. 2009. *Bromus tectorum* (Cheatgrass, Downy Brome). Given to the Alaska Wildlife Society, University of Alaska, Fairbanks, AK, April 8, 2009 (~30 audience)	Delivered
	(12) Ecological Restoration in the Shrub-Steppe. Given to the Center for Urban Horticulture, University of Washington, Oct. 24, 2007 (~30 audience)	Delivered
	(13) Link, S. O., R. Hill, R. Cruz, B. Harper, and S. Simmons. 2007. Fire Risk And Ecological Restoration In The Shrub-Steppe. Joint Washington and Oregon Chapters Wildlife Society Conference, Pendleton, OR., April, 10-13.	Delivered
	(14) Link, S. O., R. Cruz, S. Simmons, and B. Harper. 2007. Collaboration Between The Confederated Tribes Of The Umatilla Indian Reservation (CTUIR) And Washington State University (WSU) For Native Plant And Ecological Restoration Research. Some Assembly Required: Preserving Nature in a Fragmented Landscape. 34th Natural Areas Conference. Cleveland, OH Oct. 9-12.	Delivered
	(15) Link, S. O., R. Cruz, S. Simmons, and B. Harper. 2007. Formation of a Cooperative to conduct research on native plants and restore damaged ecosystems. Rethinking Protected Areas in a Changing World, George Wright Society, St.	Delivered

	Paul, MN. April 16 – 20. (16) Link, S. O., R. Cruz, S. Simmons, and B. Harper. 2007. Formation of a Cooperative to conduct research on native plants and restore damaged ecosystems. Society for Ecological Restoration Northwest Chapter Conference, Seattle, Sept. 25 (17) Link, S. O., R. Cruz, S. Simmons, and B. Harper. 2006. Ecological restoration and the control of invasives in the Columbia Basin and the Hanford Site with emphasis on collaboration between the CTUIR and WSU. National Tribal Invasive Species Conference, Sparks, NV, Nov. 7-9.	Delivered Delivered
Publications: Peer review	(1) Link, S. O., W. H. Mast, and R. W. Hill. 2006. Shrub- steppe Restoration in D. Apostol and M. Sinclair, Eds. Restoring the Pacific Northwest: The art and science of ecological restoration in Cascadia. Pp. 216-240. Island Press, Washington DC. (2) Link, S. O., C. Keeler, R. W. Hill, and E. Hagen. 2006. *Bromus tectorum* cover mapping and fire risk. International Journal of Wildland Fire 15:113-119. (3) Link, S. O. 2006. Natural areas. Natural Areas Journal 26:113. (4) Link, S. O. 2013. Management of fuel loading in the shrub-steppe: Responses six, seven, and eight years after treatments. To be submitted to Invasive Plant Science and Management or Range Ecology & Management (see final report) (5) Link, S. O. and R. W. Hill. 2013. Fire risk of restored shrub-steppe plant communities. (In review for resubmission to the International Journal of Wildland Fire) (6) Link, S. O. and R. W. Hill. 2013. Effects of herbicides on a shrub-steppe plant community after a prescribed fire. (in review for submission to Ecological Applications) (7) Link, S. O. and R. W. Hill. 2013. The Use of Herbicides for Native Bunchgrass Establishment after Fire in the Shrub-steppe (in review for submission to the Journal of Applied Ecology.	Delivered Delivered Delivered In review – see final report In review In review In review

IX. Literature Cited

Beran, D. D., R. E. Gaussoin, and R. A. Masters. 1999a. Native wildflower establishment with imidazolinone herbicides. Hortscience 34:283-286.

Beran D.D., Masters R.A., Gaussoin R.E. 1999b. Grassland Legume Establishment with Imazethapyr and imazapic. Agronomy Journal 91:592-596.

Beran, D. D., R. A. Masters, R. E. Gaussoin, and F. Rivas-Pantoja. 2000. Establishment of big bluestem and Illinois bundleflower mixtures with imazapic and imazethapyr. Agronomy Journal 92:460-465.

Bonham, C. D. 1989. Measurements for Terrestrial Vegetation. John Wiley & Sons, New York.

Briske D.D. 1991. Developmental Morphology and Physiology of Grasses, Eds. R. K. Heitschmidt and J. W. Stuth, Grazing Management An Ecological Perspective, Timber Press. pp. 264.

Bussler B.H., Maxwell B.D., Puettmann K.J. 1995. Using Plant Volume To Quantify Interference in Corn (*Zea mays*) Neighborhoods. Weed Science, 43:586-594.

Carlson J.R., Barkworth M.E. 1997. *Elymus wawawaiensis*: A species hitherto confused with *Pseudoroegneria spicata* (Triticeae, Poaceae). Phytologia 83:312-330.

Corbin J.D., D'Antonio C.M. 2004. Competition between Native Perennial and Exotic Annual Grasses: Implications for an Historical Invasion. Ecology 85:1273-1283.

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.

Elseroad A.C. and Rudd N.T. 2011. Can imazapic Increase Native Species Abundance in Cheatgrass (Bromus tectorum) Invaded Native Plant Communities? Rangeland Ecology & Management 64:641-648.

Hoitink D.J., Burk K.W., J. V. Ramsdell J., Shaw W.J. 2005. Hanford Site Climatological Summary 2004 with Historical Data, Pacific Northwest National Laboratory, PNNL-15160, Richland, Washington.

Hosten P.B., West N.E. 1994. Cheatgrass dynamics following wildfire on a sagebrush semidesert site in central Utah. In: Monsen, S. B., Kitchen, S. G., comps. Proceedings—ecology and management of annual rangelands; 1992 May 18–22; Boise, ID. Gen. Tech. Rep. INT-GTR-313. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. Pp 56–62.

Humphrey L.D., Pyke D.A. 1998. Demographic and Growth Responses of a Guerrilla and a Phalanx Perennial Grass in Competitive Mixtures. Journal of Ecology 86:854-865.

Klomp G.J., A. C. Hull J. 1972. Methods for seeding three perennial wheatgrass on cheatgrass ranges in southern Idaho. J. Range Manage 25:266-268.

Link, S. O., R. W. Hill, and E. M. Hagen. 2005. Management of Fuel Loading in the Shrub-steppe. Final report to the Joint Fire Science Program (112 pages). http://www.tricity.wsu.edu/shrub_steppe/fire_publications.htm.

Link, S. O., C. W. Keeler, R. W. Hill, and E. Hagen. 2006a. *Bromus tectorum* cover mapping and fire risk. International Journal of Wildland Fire 15:113-119.

Link, S. O., W. H. Mast, and R. W. Hill. 2006b. Shrub-steppe Restoration. Pp. 216-240 *in* D. Apostol and M. Sinclair, Eds. Restoring the Pacific Northwest: The art and science of ecological restoration in Cascadia. Island Press, Washington DC.
MacNally, R., and E. Fleishman. 2004. A successful predictive model of species richness based on indicator species. Conservation Biology 18:646-654.
Miller, J. H., R. S. Boyd, and M. B. Edwards. 1999. Floristic diversity, stand structure, and composition 11 years after herbicide site preparation. Canadian Journal of Forest Research 29:1073-1083.
Morris C., Monaco T.A., Rigby C.W. 2009. Variable Impacts of imazapic Rate on Downy Brome (Bromus tectorum) and Seeded Species in Two Rangeland Communities. Invasive Plant Science and Management 2:110-119.
Monsen, S. B., R. Stevens, and N. L. Shaw. 2004a. Restoring western ranges and wildlands. Pp. 1-294. Gen. Tech. Rep. RMRS-GTR-136-vol-1, Ft. Collins, CO.
Monsen S., Stevens R., Shaw N.L. 2004b. Restoring western ranges and wildlands. Gen. Tech. Rep. RMRS-GTR-136. Ft. Collins, CO. pp. 295-698.
Nelson D.L. 2004. Plant Pathology and Managing Wildland Plant Disease Systems, in: S. B. Monsen, et al. (Eds.), Restoring Western Ranges and Wildlands, USDA Forest Service Gen. Tech. Rep. RMRS-GTR-136. pp. 181-192.
O'Connor T.G. 1991. Local Extinction in Perennial Grasslands: A Life-History Approach. The American Naturalist 137:753-773.
Ogle D.G., Stannard M., Jone T.A. 2008. SNAKE RIVER WHEATGRASS Elymus wawawaiensis J. Carlson & M. Barkworth Plant Symbol = ELWA2, USDA, NRCS, Boise, ID. pp. 4.
Ott J.E., McArthur E.D., Sanderson S.C. 2001. Plant Community Dynamics of Burned and Unburned Sagebrush and Pinyon-Juniper Vegetation in West-Central Utah, In: McArthur, E. Durant, Fairbanks, Daniel J., comps, 2001. Shrubland ecosystem genetics and biodiversity: proceedings; 2000 June 13–15; Provo, UT. Proc. RMRS-P-21. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 177-191.
Owen S.M., C.H. Sieg, Gehring C.A. 2011. Rehabilitating Downy Brome (*Bromus tectorum*)–Invaded Shrublands Using imazapic and Seeding with Native Shrubs. Invasive Plant Science and Management 4:223–233.
Robertson, J. H., J. R. E. Eckert, and A. T. Bleak. 1966. Responses of grasses seeded in an Artemisia tridentata habitat in Nevada. Ecology 47:187-194.
SAS Institute. 2010. JMP Version 9. SAS Institute Inc, Cary.
Shinn S.L., Thill D.C. 2004. Tolerance of several perennial grasses to imazapic. Weed Technology 18:60-65.
Steele, R. B. D., and J. H. Torrie. 1960. Principals and Procedures of Statistics. McGraw-Hill, New York.
Stone W.A., Thorp J.M., Gifford O.P., Hoitink D.J. 1983. Climatological Summary for the Hanford Area., Pacific Northwest Laboratory, PNL-4622, Richland, Washington.
Thompson T.W., Roundy B.A., McArthur E.D., Jessop B.D., Waldron B., Davis J.N. 2006. Fire Rehabilitation Using Native and Introduced Species: A Landscape Trial. Rangeland Ecology & Management 59:237-248.

Ward A., G. Gee, S. Link, C. Wittreich, G. Berlin, Leary K. 2010. Quest for the Perfect Cap: The Prototype Hanford Barrier 15 Years Later. WM 2010 Conference Proceedings, Phoenix, AZ.
Weiner J. (1982) A neighborhood model of annual-plant interference. Ecology 63:1237-1241.
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.
Whitson T.D., Koch D.W. 1998. Control of downy brome (Bromus tectorum) with herbicides and perennial grass competition. Weed Technology 12:391-396.
Young J.A., Evans R.A. 1978. Population Dynamics After Wildfires. Journal of Range Management 31:283-289.