Fire risk of restored shrub-steppe plant communities

Steven O. Link[A] and ***Randal W. Hill***[B]

[A] School of Biological Sciences, Washington State University Tri-Cities, 2710 University Drive, Richland, WA, 99352, USA.
Corresponding author. Telephone: +1 509 948 0054; fax: +1 509 372 7219; email: slink@tricity.wsu.edu
[B] Columbia National Wildlife Refuge, PO Draw F, Othello, WA 99344, USA. Telephone: +1 509 488 2668; fax: +1 509 488-0705; email: Randy_Hill@fws.gov

Abstract. Fire frequency and size in western North America has increased with increasing cover of *Bromus tectorum*, an invasive alien annual grass. We determined the fire risk of shrub-steppe ecosystems restored with large bunchgrasses eight and 18 years after drill seeding. Fire risk of two communities eight years after establishment of large bunchgrasses was 76 and 78% while fire risk was 66% in a community 18 years after restoration. After 18 years, *B. tectorum* cover was 2.8 ± 2.1 % with *Elymus wawawaiensis* (large bunchgrass) density of 2.77 plants m^{-2}. Fire risk of communities dominated by small bunchgrasses (*Poa secunda*, Sandberg's bluegrass) was not different from fire risk in communities dominated by large bunchgrasses. Prescribed burns, herbicides, seed, and drill seeding are the primary costs of reducing fire risk in the shrub-steppe where drill seeding is possible. The savings associated with reducing fire risk from 100% to 66% may be enough to cover the costs of restoring shrub-steppe ecosystems. This is especially true if the savings are computed over a number of years.

Additional keywords: soil cryptogams, perennials, ignition, restoration.

Introduction

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

It is possible to reduce the cover of *B. tectorum* by restoring competitive bunchgrasses (Whitson and Koch 1998). When this is done, fire risk is reduced because there is less continuous *B. tectorum* fuel, and greater open soil plus soil cryptogam cover. The relationship between fire risk and cover of *B. tectorum*, soil and soil cryptogams, native perennials (primarily the small bunchgrass, *Poa secunda* or Sandberg's bluegrass), and litter was determined in an ecosystem similar to that of this study (Link et al. 2005). When cover of soil and soil cryptogams was 32% and *B. tectorum* 12%, fire risk was 46%. In contrast, when cover of soil and soil cryptogams was 4% and *B. tectorum* 62%, fire risk was 100% (Link et al. 2005). While it is known that high *B. tectorum* cover leads to increased fire frequency (Whisenant 1990) there has been no work describing the reduction of fire risk of restored communities. In this study, we determined the risk of fire in shrub-steppe ecosystems eight and 18 years after restoration with large competitive bunchgrasses.

In addition, because we determined the risk of fire in communities dominated by large bunchgrasses we can compare the risk with that occurring in communities dominated by small bunchgrasses (Link et al. 2005). There is a positive correlation between fine fuel biomass and maximum temperatures (Whelan 1995) which should increase the likelihood that fire will spread to nearby fuels. In the tests with the small bunchgrass (Link et al. 2005) sometimes a fire would not sustain itself in the light fuels. We test the hypothesis that fire risk will be greater in communities with large bunchgrasses than in communities with small bunchgrass.

Knowledge of the reduction in fire risk caused by restoring ecosystems should have an effect on insurance rates and lead to reductions in costs associated with preparation for fires and fire fighting. Whisenant (1990) notes that while it is possible to establish competitive bunchgrasses that can reduce fire frequency, little progress in restoring competitive bunchgrasses has been made because fiscal restraints prevent land management agencies from making significant progress. Understanding how much fire risk can be reduced through restoration should aid in budget justification. Our objective was to determine the fire risk of communities that are at differing stages of restoration. This knowledge will aid land managers who need to demonstrate the value of reducing fire risk and how long it may take to significantly reduce fire risk. To compare the restored communities, we collected information on species richness, cover, large bunchgrass density, and biomass.

Methods

Study areas

Study areas (Hampton Lake, Lake Marie, and Scaup Lake) are on the Columbia National Wildlife Refuge (CNWR) in Grant County, Washington. The CNWR is semi-arid with most precipitation (annual average 203 mm) in the fall and winter. Slopes of study areas were between 0 and 10°. Aspect of Hampton Lake was about 270°, Lake Marie 180°, and Scaup Lake 90°.

Plant communities at the CNWR have been dominated by *Artemisia tridentata*, *Pseudoroegneria spicata*, and *P. secunda* and are classified as an *A. tridentata – P. spicata* association (Daubenmire 1970). Many areas now are infested with *B. tectorum*.

Restoration of the study areas

The Hampton Lake field burned in a wildfire on July 6, 1986. The area was dominated by *B. tectorum* (CNWR 1986). The soil is an Ephrata fine sandy loam. In October 1986, the 41 ha area was drill seeded with *Elymus wawawaiensis*, *Achillea millifolium*, *Balsamorhiza sagittata*, *Linum lewisii*, *Eriogonum umbellatum*, and *Sanguisorba minor*. There is no record of herbicide use (CNWR 1986), and there have been no further treatments.

The Lake Marie field had been farmed (using irrigation) as recently as the early 1980s. The 16 ha field was dominated by *B. tectorum*, but had sparse *Agropyron cristatum*, evidence of a conversion from agriculture to perennial grass. The soil is a Malaga gravelly sandy loam. The field was sprayed with a mixture of glyphosate (561 g ha^{-1}), ammonium sulfate (20 g l^{-1} water), and surfactant in mid-November 1996. The day after spraying, snow fell and remained until late winter. Bunchgrasses [*Elymus wawawaiensis*, *Elymus lanceolatus*, *Leymus cinereus*, and *Poa secunda* (Sherman big bluegrass)] were drill seeded in early March. The field was sprayed with a light rate of glyphosate a week later after germination of *B. tectorum*, but before any drilled seed had emerged. Additional *B. tectorum* and *Sisymbrium altissimum* germinated after the last glyphosate treatment. Drill seeded grasses were well established in the year after seeding.

The Scaup Lake field had been farmed using irrigation until 1967. The soil is an Ephrata fine sandy loam. The 16 ha field was drill seeded with *A. cristatum* in spring 1968 and became infested with *B. tectorum*. Establishment of *A. cristatum* was only partly successful. Technically, *A. cristatum* is not a restoration grass because it is not native to North America, but is lumped with other native perennial grasses used for restoration. *Pseudoroegneria spicata* was planted in October 1970 and reported successful the next year. No plants were observed in the study plots in 2004. No further treatments occurred until a 1996 prescribed burn. In March 1997, the field was sprayed with mixture of glyphosate (631 g ha^{-1}), ammonium sulphate (20 g l^{-1} water), and surfactant to control *B. tectorum*. This treatment had no long-term effect on existing large bunchgrasses or *Achillea millifolium*, but eliminated germinated *B. tectorum*. The field was drill seeded with *E. wawawaiensis and P. secunda* (Sherman big bluegrass) in early March 1997.

Experimental design

In each area, 50 plots were used to determine fire risk. Plots were located in an area that was homogeneous and representative of the larger area that had been restored. Each plot was 10 m on a side. Each replicate consists of 10 plots that were randomly assigned from the population. Thus, there are 5 replicates in each of the three restored communities.

Seven plots, spaced 5 m apart, were aligned along 100 m bands. Seven bands were aligned about 10 m apart. Another plot was located at the end of one band. This design yielded 50 plots with spaces between plots for fire crews and trucks to control the test fires.

Cover determination

Percent cover of each vascular plant species, bare soil, soil cryptogams, and litter was determined in late August and early September 2004. Cover was determined using a tape (Bonham 1989; Elmore et al. 2003; Link et al. 2005) and identifying the first observed (tallest) cover type at each 0.25m hash mark on the tape. The tape was stretched tightly between two pieces of rebar metal stakes in the ground at the ends of any transect. One tape transect was used in each of seven sample plots resulting in 41 observations for each plot. Sample plots were randomly selected from the 50 plots.

Bunchgrass density

The density of bunchgrasses was determined in 7 randomly located plots in each of the three areas. All bunchgrasses except *P. secunda* were counted by species in a 3 m by 10 m band within most plots. At Hampton Lake, a 6 m by 10 m band was used in three plots.

Biomass determination

Biomass was determined in three plots at each site. Plots were 1 m by 2 m. All above ground biomass was clipped and separated into restored bunchgrasses, standing dead including the small bunchgrass, *P. secunda*, and litter. Biomass was dried in a convection oven at 50°C for seven days and weighed.

Fire risk assessment

We determined the probability of a sustained fire by trying to ignite fires in the plots of any replicate and determining the number of sustainable fires as in Link et al. (2005). We define a sustainable fire to be one that attains steady state in an area of about $100m^2$. Fires were ignited on the upwind side of a plot. The point of ignition was randomly chosen along a tape placed 90° to the wind direction and stretched across the plot. Ignition attempts were only conducted at wind speed less than 16km h^{-1}. Wind speed was measured using a digital handheld anemometer at chest height before, in the middle, and at the end of a burn. A commercial cigarette lighter was used to attempt to ignite the point near the ground.

Fuel moisture was determined before burns by harvesting three to five paper bags of about 50 g of bunchgrasses. Portions of bunchgrasses were harvested from the base and included the entire shoot. Wet biomass was determined immediately after clipping in the field using an Ohaus Scout electronic scale with 0.1 g resolution. Biomass was dried in a convection oven at 50°C for seven days and weighed again to compute percent fuel moisture. Fuel moisture ranged from 10.5 ± 1.5 to 12.1 ± 2.4% at the three sites.

Table 1. Characteristics of the three study areas for experimental fires in 2004 (n = 50).

Study areas	**Date**	**Time**	**Wind speed (km h^{-1})**	**Air temperature (°C)**	**Relative humidity (%)**
Hampton Lake	9-23	14:00 – 16:30	3.9 ± 0.3	27.1 ± 0.2	44.8 ± 0.3
Lake Marie	9-27	13:45 – 15:15	5.9 ± 0.4	31.3 ± 0.1	34.5 ± 0.7
Scaup Lake	9-30	12:00 – 15:00	6.7 ± 0.3	25.6 ± 0.1	30.5 ± 0.1

Data analysis

Analyses were done using JMP version 5.0 software (SAS Institute 2002). Percentage data were transformed using arcsin $\sqrt{p}$, where p is a proportion, before analysis (Steele and Torrie 1960). Multiple comparisons were made on transformed data using Tukey's HSD test. Means and error terms are presented using percent data for interpretation. Error terms are one standard error of the mean. Statistical significance is set at $\alpha = 0.05$.

Results

The three restored sites varied with respect to species composition and cover characteristics (Table 2). The large bunchgrasses used for restoration were all present eight and 18 years after drill seeding. Species richness was greater at Hampton Lake (18) than at Lake Marie (13) or Scaup Lake (12). The species at Hampton Lake were 83% native while only 62% and 66% were native at Lake Marie and Scaup Lake, respectively. Fifty percent of the species at Hampton

Lake were perennial compared with only 31% and 33% at Lake Marie and Scaup Lake, respectively.

There was significantly greater *P. secunda* (Sandberg's bluegrass) and, significantly, less *B. tectorum* percent cover at Hampton Lake than at Lake Marie or Scaup Lake, which were the same (Table 2). Percent cover of native species was significantly greater at Hampton Lake than at Lake Marie or Scaup Lake, which were the same (Table 2). Percent cover of alien species at Hampton Lake was significantly lower than at Lake Marie or Scaup Lake, which were the same (Table 2). Percent cover of litter at Hampton Lake was significantly lower than at Lake Marie or Scaup Lake and cover was significantly greater at Lake Marie than at Scaup Lake (Table 2). Fire risk was significantly less than 100% at all three sites.

Table 2. Presence (+) and percent cover ± 1 se (n = 7) of species and other categories plus percent fire risk ± 1 se (n = 5) at three restoration sites at the Columbia National Wildlife Refuge. Names are from Hitchcock and Cronquist (1976) with the most current names obtained from (USDA NRCS 2003).

Species, cover categories, fire risk	Native, Alien; Perennial, Annual	Hampton Lake 1986	Lake Marie 1997	Scaup Lake 1997
Boraginaceae				
Amsinckia tessellata	N; A		5.9 ± 2.1	3.8 ± 1.7
Lappula redowskii	N; A	+		
Caryophyllaceae				
Holosteum umbellatum	A; A	+	+	
Chenopodiaceae				
Chenopodium leptophyllum	N; A			0.3 ± 0.3
Salsola kali				0.3 ± 0.3
Compositae				
Achillea millifolium	N; P	+		
Agoseris heterophylla	N; A	+		+
Artemisia tridentata	N; P	+		
Erigeron pumilus	N; P	+		
Lactuca serriola	A; A		+	
Stephanomeria paniculata	N; A	+		
Cruciferae				
Descurainia pinnata	N; A		+	
Lepidium perfoliatum	A; A		+	
Sisymbrium altissimum	A; A	+	1.7 ± 0.9	0.3 ± 0.3
Graminae				
Agropyron cristatum	A; P			8.0 ± 2.9
Bromus tectorum	A; A	2.8 ± 2.1	18.5 ± 2.9	5.6 ± 2.1
Elymus wawawaiensis (Secar)	N; P	48.5 ± 5.5	32.8 ± 3.4	6.6 ± 2.5
Hesperostipa comata	N; P	+		
Leymus cinereus	N; P		6.3 ± 2.0	
Poa secunda (Sandberg's bluegrass)	N; P	16.4 ± 3.2	0.7 ± 0.4	3.1 ± 2.4
Poa secunda (Sherman big bluegrass)	N; P		1.4 ± 0.7	9.8 ± 4.4

Vulpia microstachys	N; A	1.0 ± 0.7	+	1.0 ± 0.5
Leguminosae				
Astragalus cf. lentiginosus	N; P	+		
Onagraceae				
Epilobium paniculatum	N; A	1.0 ± 1.0	0.7 ± 0.4	12.2 ± 1.4
Plantaginaceae				
Plantago patagonica	N; A	+		
Polemoniaceae				
Phlox longifolia	N; P	1.0 ± 0.7		
Umbelliferae				
Lomatium macrocarpum	N; P	+		
Natives		68.2 ± 4.9	47.7 ± 2.1	36.9 ± 4.7
Aliens		2.8 ± 2.1	18.5 ± 2.9	13.6 ± 1.8
Perennial grasses		64.9 ± 5.6	41.1 ± 3.2	27.5 ± 4.2
Litter		9.4 ± 2.7	17.8 ± 1.2	33.1 ± 3.8
Soil + soil cryptogams		23.7 ± 1.6	19.5 ± 4.7	15.0 ± 3.7
Fire risk		66 ± 5.1	78 ± 4.5	76 ± 6.8

Large bunchgrass density at Hampton Lake was significantly less than that of all large bunchgrasses (4.26 ± 0.25) at Lake Marie (Table 3). Total biomass was the same at all three sites (Table 4).

Table 3. Density (plants m^{-2} ± 1 se, n = 7) of perennial grasses planted at three restoration sites at the Columbia National Wildlife Refuge.

Species	**Hampton Lake 1986**	**Lake Marie 1997**	**Scaup Lake 1997**
Elymus wawawaiensis (Secar)	2.77 ± 0.23	3.01 ± 0.29	0.91 ± 0.19
Poa secunda (Sherman big bluegrass)	0	0.83 ± 0.37	1.44 ± 0.48
Leymus cinereus	0	0.43 ± 0.09	0
Agropyron cristatum	0	0	1.40 ± 0.44

Table 4. Above ground biomass ± 1 se (n = 3) of three restored communities at the Columbia National Wildlife Refuge.

Site	**Large bunchgrass (g m^{-2})**	**Standing dead (g m^{-2})**	**Litter (g m^{-2})**	**Total biomass (g m^{-2})**
Hampton Lake	168.9 ± 52.2	7.6 ± 1.7	32.3 ± 14.2	208.7 ± 41.2
Lake Marie	139.9 ± 6.4	10.8 ± 2.5	70.9 ± 31.0	228.0 ± 32.2
Scaup Lake	101.9 ± 20.0	10.3 ± 1.6	96.4 ± 55.5	208.6 ± 70.3

Fire risk of the three restored communities dominated by large bunchgrasses is not significantly different from that of communities dominated by the small bunchgrass, *P. secunda* when examined as a function of soil and soil cryptogam cover (Fig. 1). The linear relationship ($y = 108.8 \pm 4.2 - 1.88 \pm 0.22\ x$) between mean percent fire risk (y) and mean cover of soil and soil cryptogams (x) for the combination of areas with small and large bunchgrasses is highly significant ($p < 0.0001$) with an $r^2 = 0.69$ (Fig. 1).

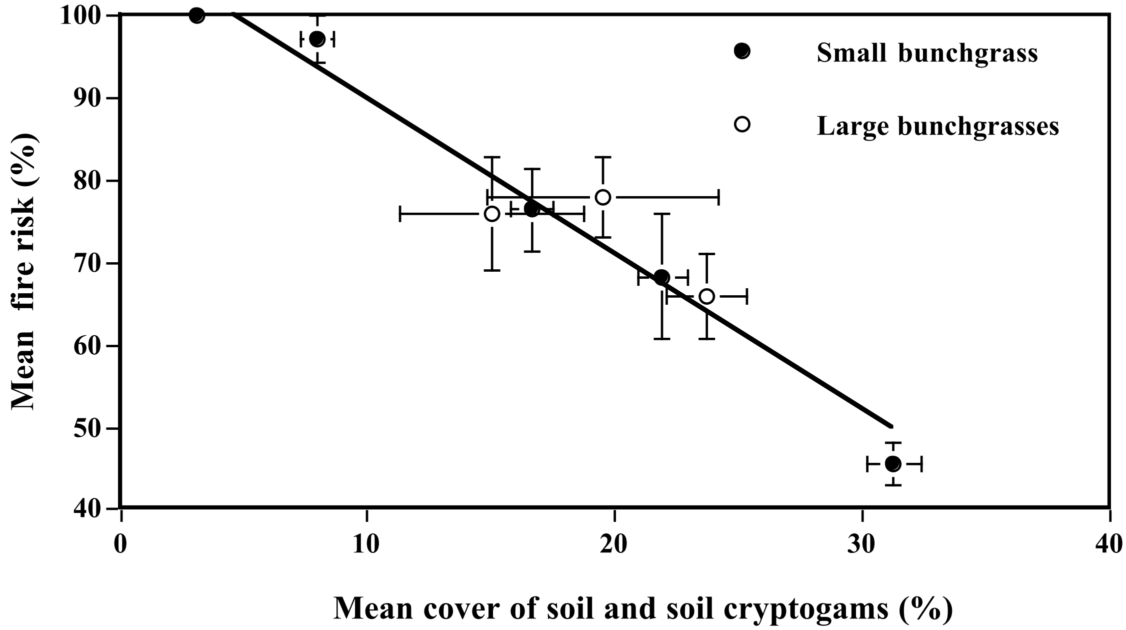

Fig. 1. Mean fire risk as a function of mean cover of soil and soil cryptogams in an area with the small bunchgrass, *P. secunda* (Link et al. 2005) compared with the three sites at the CNWR dominated by large bunchgrasses. Bars are one standard error of mean fire risk and mean cover (n = 4 for the small bunchgrass; n = 5 for mean fire risk and n = 7 for mean cover for the large bunchgrasses).

Discussion

Fire risk can be controlled in shrub-steppe ecosystems by increasing the percent cover of open soil and soil cryptogams. In the semi-arid shrub-steppe, planting large bunchgrasses can increase the amount of open soil and soil cryptogams in ecosystems dominated by *B. tectorum*, thus reducing fire risk. We discuss the dynamics of species richness, composition, cover, and fire risk for restored communities. The effect of bunchgrass size on fire risk is assessed as a function of soil and soil cryptogam cover.

The three restored fields were at differing stages of succession. Early seral communities are often dominated by annual species (Luken 1990) while late seral communities are dominated by perennials (Gross 1987) and can have high species richness (Gilbert and Anderson 1998). Disturbed fields in the shrub-steppe are often dominated by *B. tectorum* and can remain in an early seral state. Fifty years after disturbance, an old-field (Lower Snively) at the nearby Hanford Site had only nine species of which only 22% were perennials with 97% cover provided by *B. tectorum* (Rickard and Vaughan 1988). Lower Snively field does not have large bunchgrasses. The Lake Marie and Scaup Lake sites, eight years after restoration, both are more advanced than the Lower Snively field. The addition of large perennial bunchgrasses to the Lake Marie and Scaup Lake sites are likely the reason they are at a later seral stage (31 and 33%

perennial species plus 13 and 12 species, respectively). *Bromus tectorum* cover was only 18% at Lake Marie and 5.6% at Scaup Lake. The Hampton Lake site with 50% perennial species and 65.9% perennial cover after 18 years, was more similar to late successional communities that are dominated by perennial species (Gross 1987). The Hampton Lake site had higher species richness (18) than the other two sites at CNWR or the old-field at the Hanford Site, which is another indication of a later seral stage after disturbance and rehabilitation (Gilbert and Anderson 1998; Wali 1999). A similar successional pattern has been found after disturbance at a semi-arid site in North Dakota where species richness increased up to 45 years after restoration of a highly disturbed mine (Wali 1999). *Bromus tectorum* cover was only 2.8% at Hampton Lake. We conclude, under the conditions of this study, that restoration with large bunchgrasses leads to a reduction of *B. tectorum* cover.

Fire risk was less than 100% for the three restored bunchgrass communities. Risk was reduced to 76 and 78% in communities eight years after restoration and reduced to 66% in a field 18 years after restoration. It is likely that the reduction in fire risk is related to increases in the amount of bare soil and soil cryptogams (Fig. 1). Fire risk is very small in deserts where fuel is sparse and there is a high percent cover of open soil and rocks (Bond and Wilgen 1996). In a field adjacent to the Hampton Lake site, dominated by *B. tectorum*, percent cover of soil and soil cryptogams is less than 5% (Link et al. 2003a) and likely would have fire risk of 100% (Link et al. 2005). We conclude that it takes up to 18 years for soil and soil cryptogam cover to increase to about 24% after restoration with large bunchgrasses to reduce fire risk to 66%. It is likely that fire risk of 66% is close to that of shrub-steppe ecosystems dominated by large bunchgrasses without *A. tridentata* that have not been disturbed. Fire risk in undisturbed shrub-steppe ecosystems dominated by large bunchgrasses has not been determined.

The hypothesis that fire risk will be greater in communities with large bunchgrasses than in communities with small bunchgrass was false. Fire risk was strongly influenced by the amount of soil and soil cryptogam cover and not affected by bunchgrass size under the conditions of the test (Fig. 1). Given that above ground biomass of *P. secunda* is much lower, 3.3 g m^{-2} in a nearby area (Link et al. 2003b), than that of the large bunchgrasses in this study and that maximum temperatures increase with increasing fine fuel biomass (Whelan 1995) we had expected fire risk to be greater in the large bunchgrass fields. In the large bunchgrass trials, all fires were sustained when a large bunchgrass was ignited. Even though some fires were not sustained in all small bunchgrass areas (Link et al. 2005), the effect on fire risk was not strong enough to be recognized as different from the fire risk of areas with larger bunchgrasses. Given that we were not able to distinguish fire risk in areas dominated by small bunchgrasses from that of areas with large bunchgrasses we suggest that, over the range of comparable soil and soil cryptogam cover, fire risk is the same.

Conclusions

We demonstrated that fire risk is significantly reduced by introducing large competitive bunchgrasses into shrub-steppe ecosystems dominated by *B. tectorum*. Establishing *E. wawawaiensis* leads to a sustainable reduction in *B. tectorum* cover and thus a reduction in fire risk. After 18 years, *B. tectorum* cover was 2.8 ± 2.1 % with *E. wawawaiensis* density of 2.77 plants m^{-2}. Prescribed burns, herbicides, seed, and drill seeding are the primary costs of reducing fire risk in the shrub-steppe where drill seeding is possible. The savings associated with reducing fire risk from 100% to 66% may be enough to cover the costs of restoring shrub-steppe ecosystems especially if computed over a number of years.

Further research should be conducted to determine if fire risk is related to species composition or if fire risk is primarily a function of bare soil and soil cryptogam cover.

Acknowledgements

We thank Susan Nelson and Dr. Sally Simmons for suggestions that improved the manuscript. Dr. Marita Lih, Miljan Ignactic, Carson W. Keeler, Debbie Curtis, David Sexton, Jim Shultz, and fire crews of the Columbia and Mid-Columbia River National Wildlife Refuge Complex plus the Hanford Reach National Monument assisted in the field. The Joint Fire Science Program supported this work.

References

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

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

CNWR. 1986. Annual Narrative Report Calendar Year 1986. U. S. Department of Interior, Fish and Wildlife Service, Othello, WA.

D'Antonio, C. M., and P. M. Vitousek. 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global change. Annu. Rev. Ecol. Syst 23:63-87.

Daubenmire, R. 1970. Steppe vegetation of Washington, Pullman, Washington.

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.

Gilbert, O. L., and P. Anderson. 1998. Habitat Creation and Repair. Oxford University Press, Oxford.

Gross, K. L. 1987. Mechanisms of colonization and species persistence in plant communities. Pp. 173-189 *in* W. R. Jordan III, M. E. Gilpin and J. D. Aber, eds. Restoration ecology: A synthetic approach to ecological research. Cambridge University Press, Cambridge.

Hitchcock, C. L., and A. Cronquist. 1976. Flora of the Pacific Northwest. University of Washington Press, Seattle.

Knapp, P. A. 1998. Spatio-temporal patterns of large grassland fires in the Intermountain West, U.S.A. Global Ecology and Biogeography Letters 7:259-272.

Link, S. O., R. W. Hill, and E. M. Hagen. 2003a. Management of Fuel Loading in the Shrub-Steppe. http://www.tricity.wsu.edu/shrub_steppe.

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

Link, S. O., J. L. Smith, J. J. Halvorson, and J. H. Bolton. 2003b. Effect of climate change on a perennial bunchgrass and soil carbon and nitrogen pools in a semi-arid shrub-steppe ecosystem. Global Change Biology 9:1097-1105.

Luken, J. O. 1990. Directing ecological succession. Chapman and Hall, London.

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

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

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

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

SAS Institute. 2002. JMP Statistics and Graphics Guide, Version 5. SAS Institute Inc, Cary.

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

USDA NRCS. 2003. The PLANTS Database, Version 3.5 (http://plants.usda.gov). National Plant Data Center, Baton Rouge.

Wali, M. K. 1999. Ecological succession and the rehabilitation of disturbed terrestrial ecosystems. Plant and Soil 213:195-220.

Whelan, R. J. 1995. The ecology of fire. Cambridge University Press, Cambridge.
Whisenant, S. G. 1990. Changing fire frequencies on Idaho's Snake River plains: ecological and management implication. USDA Forest Service Intermountain Research Station General Technical Report: INT-276:4-10
Whitson, T. D., and D. W. Koch. 1998. Control of downy brome (*Bromus tectorum*) with herbicides and perennial grass competition. Weed Technology 12:391-396.