
Regional Strategies for Restoring Invaded Prairies

**Final technical report
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PREFACE

This report is the result of a cooperative project between the Institute for Applied Ecology (IAE) and The Nature Conservancy (TNC). IAE is a non-profit organization dedicated to natural resource conservation, restoration, research, and education. Our mission is to conserve native species and habitats through restoration, research and education.

Cover photo: Amanda Stanley

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Data sharing

Raw and summarized data from this project is available upon request to project partners and to the broad community of restoration practitioners and ecologists working in this region. Please contact the project coordinator if you have specific questions unanswered in this report

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SUMMARY

Background and approach: Invasive plants, especially non-native perennial grasses, are a critical threat to remnant prairies and oak savannas in the Pacific Northwest. We evaluated the effectiveness of restoration treatments designed to 1) reduce target exotic weeds with minimal non-target impacts and 2) increase native species diversity and abundance. In cooperation with numerous partners, the Institute for Applied Ecology and The Nature Conservancy conducted a 5-year study at 10 sites along a 500-km latitudinal gradient from the Willamette Valley, OR to Vancouver Island, BC. Our manager-recommended treatment combinations included the following components: summer and fall mowing, grass-specific and broad-spectrum herbicide, and fall burning. All treatment combinations were crossed with native seed addition. Each combination was created to target factors likely to limit restoration success, including extreme exotic grass cover, litter accumulation, and native seed limitation. Where possible, we also applied what appeared to be the most promising treatment combination over a large area (100 x 100 m) to assess the scalability of results. Results from small-scale studies may differ substantially when applied at large scales due to edge effects in small plots (e.g., seed inputs from untreated areas), community level effects (e.g., impacts from herbivores), or spatial heterogeneity.

Results: After 5 years, we found that the most disturbance-intensive treatment combination (sethoxydim, burning, and post-fire glyphosate) led to reduced abundance of exotic grasses and forbs without causing a decline in native species. Sethoxydim combined with fall mowing reduced exotic grasses and increased native plant abundance. In all cases, disturbance treatments reduced exotic cover to varying degrees but had no positive impact on native diversity; only seed addition increased native species richness. Results from the large treatment areas were complicated due to difficulties with treatment application and timing, but generally reflect the results from the small-scale experiments. Our results show that restoration of degraded grasslands is most successful when it employs a variety of strategies applied in combination over several years, and where the type, timing, and number of treatments are carefully chosen based on a thorough understanding of limiting conditions, species biology, and grassland ecology.

Management recommendations: As expected, we found there was no 'silver bullet.' While some treatment combinations led to large improvements in weed control and native diversity and abundance, the degree of success varied across sites. Where invasive grasses are the most pressing problem, we recommend the use of grass-specific herbicides as highly effective with minimal non-target effects on native forbs and some native grasses. Fire is a useful tool for preparing a site for seeding, but may need to be followed closely with a broad spectrum herbicide to control rapidly resprouting weeds. Careful timing of post-fire herbicide avoids later-sprouting natives. At all sites, we recommend seed addition to enhance native diversity and abundance, as our data show even relatively high quality sites show strong seed limitation. Mowing is ineffective at reducing weed abundance, and can negatively impact some natives depending on timing. While mowing did reduce thatch and increase light penetration, it did not increase bare soil leading to low seedling success. If fire is not an option, we recommend testing mowing in combination with treatments to reduce moss and increase bare soil.

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INTRODUCTION

Invasive plants, especially non-native perennial grasses, pose one of the most critical threats to protected prairies in the Willamette Valley/Puget Trough/Georgia Basin Ecoregion (WPG – Figure 1). The remaining prairies, often small and fragmented, are among the most endangered ecosystems in North America and support many imperiled species (Noss and others 1995; Dunn and Ewing 1997; Floberg and others 2004). Invasive species reduce native diversity and alter vegetation structure, fire regimes, soil characteristics, and faunal diversity. Many of these invaded remnant sites still retain a desirable native biota, presenting a particularly difficult restoration challenge. Part of the dilemma is how to selectively remove exotics without causing damage to natives. Native and exotic species often share many traits (e.g., phenology, physiology, degree of susceptibility to grazing or fire), and management actions that effectively control invasive species often impact many native species as well (Smith and Knapp 1999, Sheley and Krueger-Mangold 2003). Additionally, our current knowledge regarding the effectiveness of restoration techniques is largely anecdotal or based on results from only a few site specific studies.

While competition from non-native plants limits or prohibits native species from expanding or re-establishing in invaded habitats (D'Antonio and Kark 2002, Levine et al. 2003), the ability of the native plant community to rebound after exotic removal remains unknown. Seed availability is a strong constraint on native species establishment and increase (Foster and Tilman 2003, Clark et al. 2007), and thus native seed addition may be required in these habitats to improve native diversity and abundance.

Objectives

To address these problems, The Nature Conservancy worked jointly with the Institute for Applied Ecology and numerous partners from 2005 – 2010 to conduct a long-term, ecoregion-wide study of prairie restoration methods. Our objectives were to:

1. Improve strategies for controlling herbaceous non-native weeds, particularly perennial grasses, while maintaining or enhancing the abundance and diversity of native plants, and
2. Generalize these results to develop strategies that can be applied throughout the ecoregion.

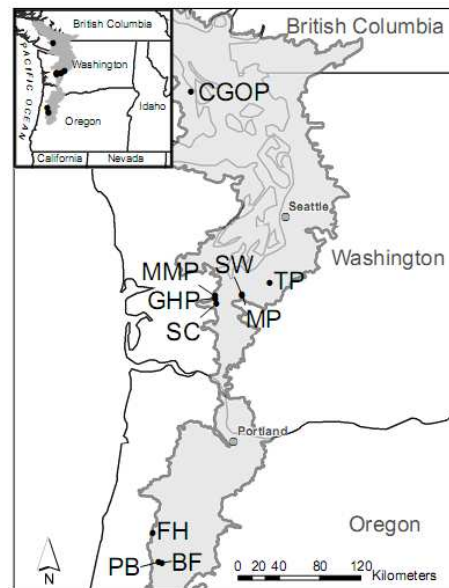


Figure 1. Map of study sites and WPG ecoregion. (CGOP = Cowichan Garry Oak Preserve; SW = South Weir Prairie; MMP = Mima Mounds Preserve; GHP = Glacial Heritage Preserve; SC = Scatter Creek; TP = Triangle Prairie; MP = Morgan Property; FH = Fort Hoskins; BF = Bellfountain Road; PB = Pigeon Butte).

Approach

This project was developed as a collaboration between researchers and land managers (Stanley et al. 2008). We tested multi-faceted restoration techniques for reducing invasive species abundance at ten prairie and oak savanna sites throughout these highly fragmented habitats within the WPG ecoregion (Fig. 1, Noss et al. 1995, Dunn and Ewing 1997). At each site, we tested disturbance treatments (fire, mowing, herbicide) and native seed addition in a replicated experiment, using small-scale plots (5 x 5 m). Treatment combinations were developed collaboratively by participating land managers and scientists.

While most scientists test treatments alone or in limited combinations in a factorial design, we decided to test *treatment combinations* developed in an adaptive management framework, as this would be most applicable to on-the-ground management. Our treatment combinations included grass-specific herbicide (sethoxydim), broad-spectrum herbicide (glyphosate), fall burning, spring mowing, fall mowing, and seeding with native species.

Where possible, we also applied what appeared to be the most promising treatment combination over a large area (100 x 100 m) to assess the scalability of results. Results from small-scale studies may differ significantly when applied at large scales due to edge effects in small plots (e.g., seed inputs from untreated areas), community level effects (e.g., impacts from herbivores), or spatial heterogeneity.

In this report, we address the following specific questions:

- (1) Can combined treatments reduce the abundance of non-native grasses, and which combinations are most effective?
- (2) How do non-target species (native and non-native forbs, native grasses, and annuals) respond to treatment combinations?
- (3) Does native plant diversity and abundance increase rapidly in response to the reduction of the dominant grasses, or is seed addition required?
- (4) Are results consistent across the ecoregion, or do treatment responses vary between sites?
- (5) Are results from small-scale experimental plots consistent with results at larger scales?

Additionally, we discuss the implications of our results for broader theoretical concepts of the factors limiting the abundance and diversity of native plants. At the start of this study, our team proposed that competition and seed dispersal are the main factors limiting native plant establishment. Competition from invasive plants may suppress native species from expanding or re-establishing. Dispersal of native seed into invaded habitats may be so low that native species are unable to establish. Both of the processes may be acting at once to control the success of restoration practices. We revisit these concepts in the Conclusions section on page 26.

METHODS

Study area

We began this study in 2005 at ten sites distributed across the WPG ecoregion (Figure 1), an area of lowland prairies once burned frequently by Native Americans (Boyd 1986, Kruckeberg 1991). The ecoregion is low elevation and generally characterized by dry summers and wet, mild winters, with most vegetation growth occurring in spring and some regrowth in fall (Sinclair et al. 2006). Prairies in the WPG are challenged by fire suppression, habitat conversion, fragmentation, species invasion, and loss of diversity (Floberg et al. 2004, Dunwiddie et al. 2006). Research sites were selected in natural areas and preserves, protected and managed by various agencies and organizations (Figure 1, Table 1). Although there is considerable overlap in species composition among these prairies, they vary widely in terms of soils, climate, land use history, and degree of invasion (Floberg et al. 2004, Dunwiddie et al. 2006).

Table 1. Research site ownerships and pre-treatment diversity, measured in sampling quadrats prior to treatment in spring 2005. Relative native cover was calculated as the percentage of total vegetative cover comprised of native species.

Site	State or Province	Ownership	No. Exotic Spp.	No. Native Spp.	Relative Native Cover
Bellfountain Road	OR	Finley Wildlife Refuge, USFWS	34	34	40%
Cowichan Garry Oak Preserve	BC	Nature Conservancy of Canada	19	31	40%
Glacial Heritage Preserve	WA	Thurston County Parks	15	19	23%
Fort Hoskins Historical Park	OR	Benton County Natural Areas and Parks Dept.	24	13	36%
Mima Mounds Natural Areas Preserve	WA	WA Dept. Natural Resources	17	24	32%
Morgan Property	WA	The Nature Conservancy	12	22	28%
Pigeon Butte	OR	Finley Wildlife Refuge, USFWS	32	28	43%
Scatter Creek Wildlife Area	WA	WA Dept. Fish and Wildlife	19	28	63%
Triangle Prairie, Training Area 15	WA	Fort Lewis, US Army	23	30	48%
South Weir Prairie, Training Area 23	WA	Fort Lewis, US Army	18	21	12%

Experimental plots were located in upland prairies and oak savannas that retained at least some native species, but also had a significant presence of non-native plants, particularly invasive grasses (Table 1). Non-native perennial grasses degrade these ecosystems by their rapid growth, structural dominance, and thatch accumulation, which limit establishment and growth of native species (Sinclair et al. 2006). Prescribed fire often benefits these fire-tolerant invaders as much as native species. Grass-specific herbicides, such as sethoxydim (Poast) or fluazifop (Fusilade), provide an opportunity to target invasive perennial grasses, but most native grasses are also susceptible.

However, *Festuca roemerii* (Roemer's fescue), one of the most common matrix species in the region, is resistant to both sethoxydim (Dunwiddie and Delvin 2006) and fluazifop (Blakeley-Smith 2006).

Common dominant non-native species at the sites included the grasses *Arrhenatherum elatius* (tall oatgrass), *Agrostis capillaris* (bentgrass), *Poa pratensis* (Kentucky bluegrass), *Anthoxanthum odoratum* (sweet vernal grass), *Holcus lanatus*

(velvetgrass), *Dactylis glomerata* (orchard grass), and *Bromus hordeaceus* (soft brome), and the forbs *Hypochaeris radicata* (hairy cat's-ear) and *Leucanthemum vulgare* (ox-eye daisy). Common native species included the grasses *Festuca roemerii*, *Danthonia* spp. (oatgrass), *Bromus carinatus* (California brome), and *Elymus glaucus* (blue wild-rye), the sedges *Carex tumulicola* (split-awn sedge) and *C. inops* (long-stolon sedge), the rush *Luzula comosa* (Pacific woodrush), and the forbs *Achillea millefolium* (common yarrow), *Camassia quamash* (small camas), *Campanula rotundifolia* (bluebell bellflower), *Fragaria virginiana* (wild strawberry), *Lomatium utriculatum* (common lomatium), *Prunella vulgaris* (common selfheal), and *Ranunculus occidentalis* (western buttercup).

Experimental design and data collection: small-scale plots

Plot replication and layout

Each of our 10 sites contained a block of 20 5 x 5 m experimental plots (5 treatments x 4 replicates = 20 plots per block), with treatments randomly assigned to plots. At 8 sites, plots were laid out in a grid; at two other sites plots were laid out in other configurations. At Mima Mounds Preserve, plots were located on the tops of mounds, as vegetation communities on mounds were very different from inter-mound communities. Mounds average ca. 9 m in diameter and 2 m high. Plots at Cowichan Garry Oak Preserve were placed to avoid having oak trees within plots. All treatments except seeding were applied to the entire plot.

Table 2. Manager-recommended disturbance treatment combinations applied to 5 x 5m plots. Sethoxydim (S) is an herbicide which targets all Poaceae; however, the native grass *Festuca roemerii* (Roemer's fescue) is resistant. Glyphosate (G) is a broad spectrum herbicide and was applied two to three weeks after burning (B). Mowing (M) was conducted in late spring, fall or both. Spring treatments occurred after data collection. Native seed was added to half of each treatment unit, including controls, in fall of 2006 and fall of 2007.

Year	Season	Treatment code				
		SBG	MBG	MM	SM	Control
2005	Spring	sethoxydim	Mow	Mow	sethoxydim	
	Fall			Mow	Mow	
2006	Spring	sethoxydim		Mow	sethoxydim	
	Fall	burn + glyphosate (seed)	burn + glyphosate (seed)	Mow (seed)	Mow (seed)	(seed)
2007	Spring	sethoxydim		Mow	sethoxydim	
	Fall	(seed)	(seed)	Mow (seed)	Mow (seed)	(seed)
2008	Spring			Mow		
	Fall	burn + glyphosate	burn + glyphosate	Mow	Mow	

Treatment combinations

Treatment combinations (see Table 2) were developed in collaboration between managers and researchers. The combinations included spring application of sethoxydim (a grass-specific herbicide) to reduce abundance of exotic grasses; spring mowing to prevent seed set and reduce stored reserves invasive plants, particularly exotic perennial grasses; fall mowing to reduce thatch accumulation and cut back fall-regrowing grasses; fall burning to reduce biomass and thatch accumulation and prepare sites for re-seeding; and post-burn glyphosate (a broad-spectrum herbicide) application to reduce abundance of broad-leaf weeds. This last treatment was developed based on observations that non-native species resprout more quickly after fire than do most native species. Mowing height was 1-3" and biomass was left in place; biomass removal is impractical at large scales and managers noted that biomass appeared to decompose faster after mowing. A fully factorial design was not possible, given the number of treatments and limitations on resources and space.

Spring applications of sethoxydim were timed to plant development rather than calendar date; optimal spray time was slightly before the main target grass at each site was in the boot stage (when the seed head was still enclosed in the leaf sheath, typically mid April – mid May). Sethoxydim was applied in a 1.5% solution with a surfactant (crop oil) and marking dye. Spring mowing occurred after vegetation sampling, when the target grass was flowering. Fall mowing occurred just after the start of fall rain, when target grasses usually started a period of fall regrowth. Target mow height was 6-10 cm (2.5-4 inches). Burns occurred in early fall (September) at the end of the summer drought period when most species were dormant. Glyphosate was applied to burned plots approximately two weeks after the burn, in a 1.5% solution with surfactant (crop oil) and marking dye. Site managers assessed abundance of resprouting natives and exotics before spraying, and found that at most sites exotic species greened up quickly after the burn, while natives resprouted more slowly. At two sites with high native abundance, Cowichan Preserve and Triangle Prairie, site managers decided to spot spray to avoid native plants. This flexibility in treatment application was a compromise between researchers' desire for uniform treatments across all sites and managers' need to avoid damaging high quality prairie remnants.

Seed addition

Native seed was sown in fall 2006 and fall 2007. At each seeding, one half the plot was seeded, such that the plot was divided into four subplots: not seeded, seeded in 2006 only, seeded in 2007 only, and seeded in both years (Figure 2). The exception was the site at Cowichan preserve, where seed was sown in the same half of the plot in both years, such that the plot was divided in two sections (seeded both years, not seeded). Seed was broadcast in fall (late October – early November) followed by light raking to improve seed-soil contact. Unseeded subplots were also raked. Seeding occurred after any fall treatments (burning, herbicide, mowing).

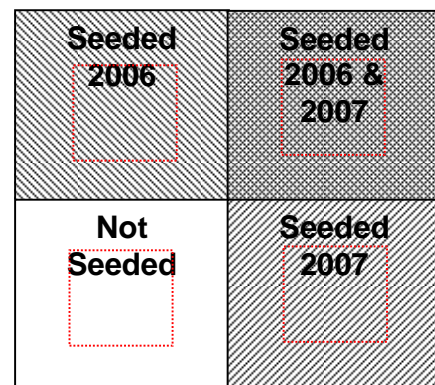


Figure 2. Subplot layout showing native seed addition in 5 x 5 m plot. Red squares indicate locations of permanent 1m² sampling quadrats.

In fall 2006 seven species that are widespread throughout the ecoregion were sown at each site, with two congeneric substitutions made based on the locally abundant species (Table 3). We chose these species to represent a range of growth forms and life histories typically found in these prairies. Native seed was either collected on site, from nearby sites, or purchased from local growers. Although we tried to standardize the quantities of seed sown at all sites, some variations occurred because seed availability was limited in certain areas (see Table 3 for seed sowing rates).

In fall 2007 we used the same seed mix with the addition of one species, *Balsamorhiza deltoidea*, at the Washington sites. Quantities varied slightly from 2006, again depending on seed availability (Table 3).

Table 3. Amount of native seed (g) added to subplots (2.5 x 5m) at each site. Our seed mix was designed to have 40% grasses/ 60% forbs (by seed number rather than seed weight), with a total seeding rate of approximately 1200 seeds/m² (12.8 lbs/acre). CGOP = Cowichan Garry Oak Preserve; SW = South Weir Prairie; MMP = Mima Mounds Preserve; GHP = Glacial Heritage Preserve; SC = Scatter Creek; TP = Triangle Prairie; MP = Morgan Property; FH = Fort Hoskins; BF = Bellfountain Road; PB = Pigeon Butte. Note: multiply grams/subplot by 0.714 to convert to lbs/acre.

Species	2006	2007	Sites
	Seeds per Subplot (g)	Seeds per Subplot (g)	
<i>Achillea millefolium</i> (per. composite)	0.36	0.44	All
<i>Balsamorhiza deltoidea</i> (tap-rooted forb)		7.1	GH,MM,SC
		6.6	MP,SW,TP
<i>Danthonia californica</i> (per. grass)	4.9	7.3	MP, TP, SW
	5.45	4.3	MMP
	8.72	8.72	BF, PB, FH, CGOP
<i>Danthonia spicata</i> (per. grass)	6.8	3.6	GHP, SC
<i>Eriophyllum lanatum</i> (per. composite)	0.47	0.81	GHP, MMP, MP, SC, TP, SW
	0.81	0.81	BF, PB, FH, CGOP
<i>Festuca roemerii</i> (per. grass)	2.93	2.93	All
<i>Lomatium nudicaule</i> (tap-rooted forb)	16.06	8.65	TP
<i>Lomatium utriculatum</i> (tap-rooted forb)	1.83	1.83	All except TP
	1.83	0.91	MM
<i>Plectritis congesta</i> (annual forb)	0.5	0.5	All
<i>Ranunculus occidentalis</i> (fibrous-rooted perennial)	2.1	2.74	GHP, MMP, MP, SC, TP, SW
	2.75	2.74	BF, PB, FH, CGOP

Data collection

Plant community composition data were collected from four 1 x 1-m permanent sampling quadrats per plot, one centered in each quarter of the plot (see Figure 2) Sampling was conducted in spring (April – early June) each year (2005-2010). We visually estimated percent cover to the nearest 1% for all vascular plant species, as well as moss, litter, and bare soil. Total cover for a plot was at least 100%, and often exceeded that if many layers of vegetation were present. We reduced the potential for observer bias by having the same lead data collector from year to year, combined with

training and frequent calibration of observations with seasonal data collectors. Species nomenclature and supplementary information (provenance, duration, etc.) followed the USDA PLANTS database (USDA and NRCS 2008) and local floras (Hitchcock and Cronquist 1973, Kozloff 2005).

Soils data were collected in 2005 at each site by Steve Griffiths and Mabelle Nelson (USDA) to characterize the soil chemistry at each site. Three 15 cm cores were taken from randomly chosen locations around the block. At four sites, 2 100 cm cores were collected from random locations (Bellfountain, Fort Hoskins, Pigeon Butte, and Scatter Creek). At the other sites, soils were too rocky or shallow to permit collection of 100 cm cores. The 100 cm cores were sliced into 5 cm increments and each slice was analyzed separately. We repeated soil sampling at the plot scale in 2006 and 2009, using 3 combine 10 cm samples, at a subset of sites.

We collected biomass data at all sites except Cowichan from 2006-2009. From 2006-2008 we collected three 10 x 100 cm strips randomly located near the experimental plots, by clipping all biomass originating in the strip to the soil surface. Biomass was sorted to forb, grass, shrub, fern, and combined litter/moss/lichen. In 2009 a 10 x 100 cm strip was collected in each experimental plot excluding Cowichan. Biomass was collected at or near peak biomass (June).

From 2006-2009 we collected data on light interception in all experimental plots excluding Cowichan using an Accupar LP-80 (Decagon Devices). Although light interception data were collected at Cowichan in 2006, these data may not be usable due to the abundance of trees (*Quercus garryana*) that blocked some direct sunlight. We took 4 paired measurements per plot (above vegetation canopy and at ground level), 1 pair at each corner. Light interception (%) was calculated as: $100 \times (\text{light above} - \text{light ground}) / \text{light above}$. We collected light measurements on cloud-free days between 10:30-2:30pm.

At each site excluding Cowichan, we collected data on litter and moss depth in each plot from 2006-2010. At 10 randomly selected locations we measured the height from the soil surface of the tallest piece of litter touching the ruler and the maximum height from the soil surface to the top of the moss layer.

Data analysis

Data were analyzed with a variety of statistical tests.

For interpreting treatment and seeding effects on cover and richness across all sites, we used linear mixed ANCOVA in SAS 9.13 (PROC MIXED; SAS Institute, 2001). Treatment and seeding were fixed interacting factors, and pretreatment data were used as the covariate. Seeding nested within treatment was used as the replicating unit. Richness data were square-root transformed and cover data were log transformed to meet assumptions of normality and homogeneity of variance. Post-hoc Tukey tests were used for testing for differences among treatment means. We used $p < 0.05$ as our criterion for significant results.

To examine site level responses (richness, cover, and species composition) we used PERMANOVA in R 2.10 (R core development team, www.cran-r.org, 2010). Again our response variable were the 2009 values with 2005 values as a covariate. We used post-hoc comparisons to test for differences between treatments, using $p < 0.05$ as our criterion for significance.

To visually examine changes in species composition, we used non-metric multidimensional scaling (NMDS) in R. To look at differences across sites, we grouped species into broad functional categories, as species turnover across sites was very high. We used only the ‘seeded in both years’ quadrats to look for the maximum change in species diversity. All four replicates at a site were averaged together. Ordination was performed on all years together. Environmental variables were fit to the ordination using a permutation test.

Experimental design and data collection: large unreplicated plots

Large plots were located at 7 sites, and varied in size and treatment application (Table 4). Site managers at each site helped select an area and the treatment combination to be tested. At all sites, the goal was to test the SBG treatment (sethoxydim-burn-glyphosate), but logistical issues prevented completion of treatments on schedule at many sites. In each plot we set up 4 randomly located transects with 5 randomly located permanent sampling quadrats, for a total of 20 quadrats/plot. Vegetation was monitored pre-treatment and then every year or every other year (Table 4). Data on vegetation, biomass, litter & moss depth were collected as described above.

Table 4. Large scale plot locations, size, treatment applications, and monitoring schedule.

Site	Size	Monitoring	2006	2007	2008	2009
Bellfountain	30 x 100m	2007, 2009			sethoxydim	sethoxydim, fall burn + glyphosate, native seeding
Glacial	100 x 100m	2007, 2009		sethoxydim	sethoxydim	
Mima	50 x 200m	2007, 2009			sethoxydim	sethoxydim
Morgan	100 x 100m	2006, 2009	sethoxydim	sethoxydim, fall burn + glyphosate, seeding with fescue	sethoxydim	
Scatter	100 x 100m	2007, 2009			sethoxydim	sethoxydim
Weir	100 x 100m	2006, 2008, 2009	sethoxydim		Fusilade + fall burn	Fusilade
Triangle	100 x 100m	2006, 2008, 2009	sethoxydim		Fusilade	Fusilade

RESULTS AND DISCUSSION – SMALL SCALE

Research question (1): Can combined treatments reduce the abundance of non-native grasses, and which combinations are most effective?

We found several of our treatment combinations were very successful at reducing non-native grasses¹. Combinations including repeat applications of sethoxydim were most effective (SBG and SM, Figure 3), and were not significantly different from each other². The combination of mowing, fall burning + glyphosate (MBG) reduced non-native grasses significantly below controls, but not as much as the sethoxydim treatments. Sethoxydim was very successful at reducing non-native grasses (Figure 3). Both SBG (sethoxydim-burn-glyphosate) and SM (sethoxydim-mow) reduced non-native grasses from a cross-site average of 16.8% cover to < 5% cover. Repeated mowing (MM) had no effect on these grasses when examined across all sites.

The efficacy of sethoxydim varied between sites, likely due to differences in

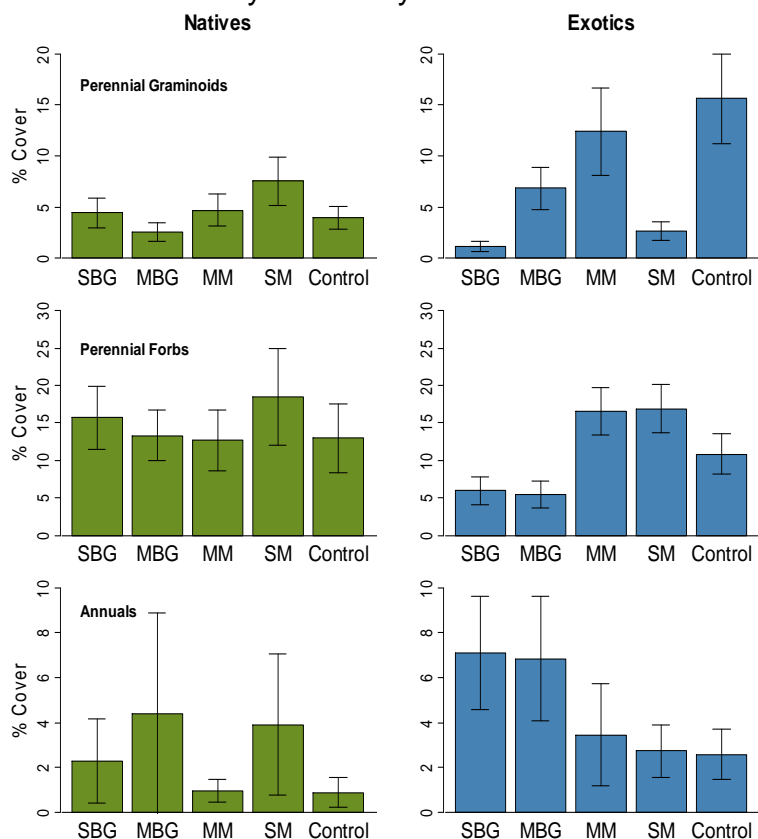


Figure 3. Functional group response to treatments in 2009. Graminoids includes grasses, sedges, and rushes; annuals includes annual grasses and forbs. Bars represent across site plot averages (all subplots combined), +/- 1 standard

application timing and target grasses (Figure 4). Sites in the Willamette Valley dominated by exotic *Agrostis* (bentgrass) species (Bellfountain and Hoskins) took longer to achieve a substantial reduction. In 2005 and 2006 in particular spraying was not at optimal times, and some of the grasses had already initiated flowering when spraying occurred. Sethoxydim was highly effective in controlling most other non-native grass species, including *Anthoxanthum odoratum*, *Arrhenatherum elatius*, and *Holcus lanatus* (see Table 5), but several applications were needed. The large amounts of thatch (average of 60% cover and 3.5 cm depth) might have played a role in this, as the thatch might have prevented the herbicide from

¹ Because native seeding had no effect on the abundance of non-native grasses, to address this question we took the average of all 4 sampling quadrats per plot.

² Data analyzed with linear mixed effects models; post-hoc Tukey tests at $p < 0.05$

reaching the entire plant. Where spraying occurred early enough, it was also effective on annual bromes (Table 5). The non-native annual *Vulpia bromoides* increased with sethoxydim treatments, either because it flowered prior to spraying or because it resisted sethoxydim like other fine-leaved fescues.

Burning and glyphosate combined with mowing (MBG) rather than sethoxydim provided some gains, depending on the target species. Grasses rebounded the second growing season after burning/glyphosate, but then were further reduced by the repeat of this treatment. *Anthoxanthum odoratum* increased in many plots with this treatment; *A. odoratum* often increases after fire, but did not resprout quickly enough after fire to be affected by the glyphosate application.

Repeat mowing was ineffective at most sites, but did appear to reduce *Arrhenatherum elatius* at Pigeon Butte and Scatter Creek, at least temporarily (Table 5). While cover of *A. elatius* was reduced by repeat mowing, plants were not killed and observations suggest they can recover quickly once mowing is stopped (J. Beall, personal communication). In 2010 these plots are not much different from controls (A. Stanley, personal observation). Mowing seemed particularly ineffective at controlling *Agrostis* spp.; at Bellfountain, *Agrostis* was the dominant non-native grass, and increased with mowing from 2005 to 2007 (Figure 4). This result is perhaps not surprising, as *Agrostis* is known to be well-adapted to mowing - hence its use as a lawn and turf grass.

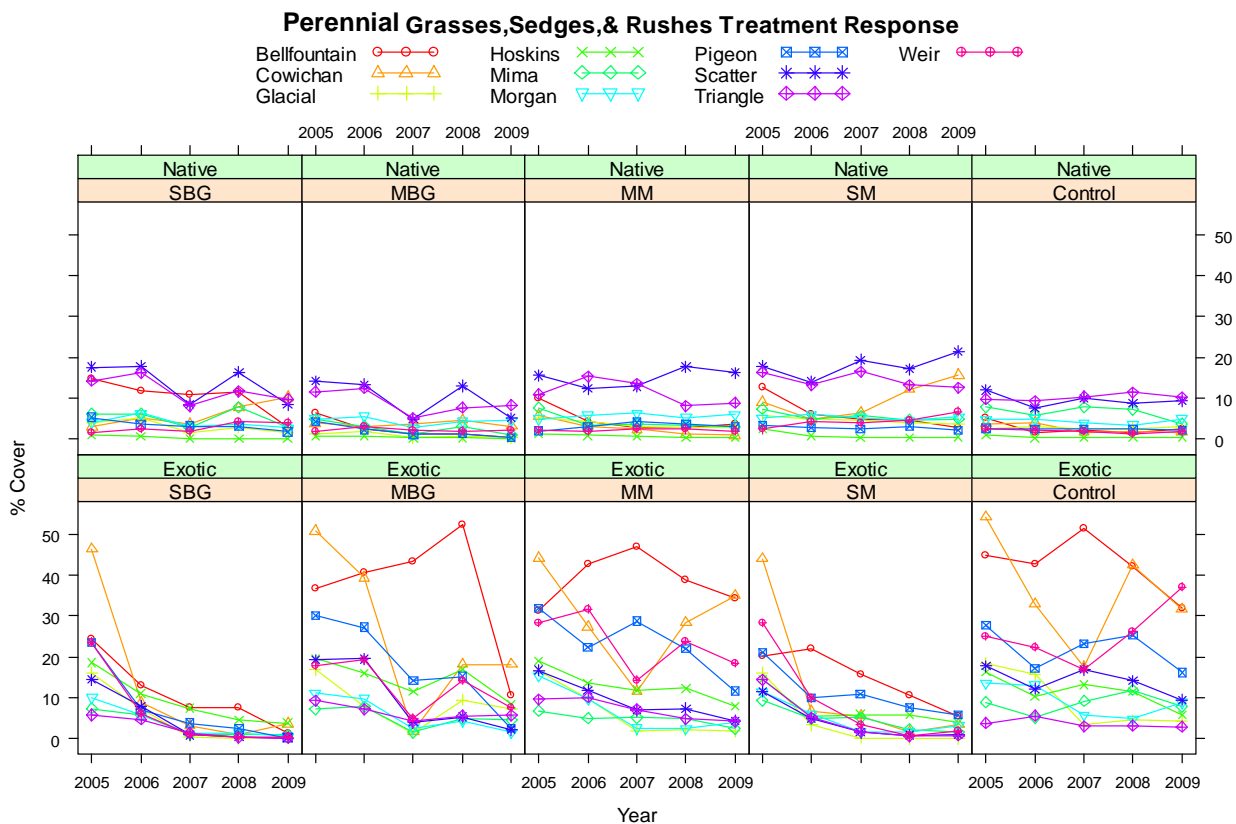


Figure 4. Response of perennial grasses, sedges, and rushes to treatments at each site.

Research question(2): How do non-target species (native grasses, native and non-native forbs and annuals) respond to treatment combinations?

Native grasses, sedges, and rushes

Across all sites, native graminoids (grasses, sedges, and rushes) either exhibited no change (SBG) or increased (SM) with sethoxydim (Figure 3)¹, indicating that any decline in sethoxydim-susceptible native grasses was offset by an increase in non-susceptible species such as *Festuca roemerii* (Blakeley-Smith 2006), the sedges *Carex tumulicola* and *C. inops*, and the wood-rush *Luzula comosa*. Native graminoids declined in treatment MBG, most likely because the dominant species at many sites, *F. roemerii*, tends to be set back by burning (Dunwiddie 2002). For example, at Scatter Creek we can see in Figure 4 that native graminoids (dominated by *F. roemerii* at this site) in the MBG combination declined in 2007 after the burn treatment, rebounded in 2008, and then declined again in 2009 following the second burn. Mowing may have been a factor as well because a significant post-fire decline was not seen in treatment SBG. *Danthonia* spp., *Elymus glaucus*, and *Bromus carinatus* all declined with sethoxydim, although *Danthonia* plants often survived several applications, perhaps due to their prostrate growth habit. Repeated mowing had no effect on native graminoids.

Perennial non-native forbs

The large reduction in dominant exotic grasses in treatments involving grass-specific herbicide (SBG and SM) could also create opportunities for other exotic species to increase. We saw increases in non-native perennial forbs in the SM treatment at many sites (Figure 5), particularly species like *Hypochaeris radicata* and *Leucanthemum vulgare* (Table 5). Post-fire glyphosate application (SBG, MBG) reduced exotic perennial forbs by almost 50% (Figure 3)². Because many exotic species resprouted more quickly after burning than most natives, this carefully timed application of a broad-spectrum herbicide was fairly selective in impacting the exotic species. Repeat mowing (MM) had little effect on non-native forbs, except at one site, Pigeon Butte, where exotic forbs increased 5x from 2005-2008 (Figure 5). Mowing of tall oat grass (*A. elatius*) increased light availability, and species such as *Daucus carota*, *Leucanthemum vulgare*, and *Vicia* spp. appeared to benefit.

Native perennial forbs

At most sites, native forbs showed modest response to treatments (Figure 5). Repeat mowing had no significant effect on native forbs at any site. At Pigeon Butte, native forbs showed the same response to mowing as non-native forbs, driven mainly by an increase in clonal species such as *Calystegia atriplicifolia*, *Sidalcea virgata*, and *Fragaria virginiana*, although this increase was non significant in 2009³. Native forb response to MBG was neutral to positive at most sites, although the increases were not large at most sites (Figure 5). Five sites showed an increase in native forbs (Cowichan,

¹ Linear mixed effects models; because seeding effect was not significant for grasses, all 4 sampling quadrats were averaged together. Post-hoc Tukey comparisons showed only SM higher than controls, and only MBG lower than controls at $p < 0.05$.

² Linear mixed effects models, significant post-hoc Tukey test. Because seeding effect was not significant for non-native forbs, all 4 sampling quadrats were averaged together.

³ Treatment effect on native forbs N.S. at Pigeon Butte using PERMANOVA

Glacial, Morgan, Scatter, and Weir), with only one site showing a decline (Bellfountain)¹. Fort Hoskins, had a non-significant decline in native forbs after burning + glyphosate, This was likely because the most abundant native species at Fort Hoskins and Bellfountain, *Fragaria virginiana*, was negatively impacted by burning. Treatments including sethoxydim (SBG, SM) also showed neutral to positive effects, with the same five sites (Cowichan, Glacial, Morgan, Scatter, and Weir) showing significant increases². Overall we see a trend where the largest increases in native forbs occur at sites with higher initial forb diversity and abundance, such as Cowichan and Scatter Creek (Figure 5).

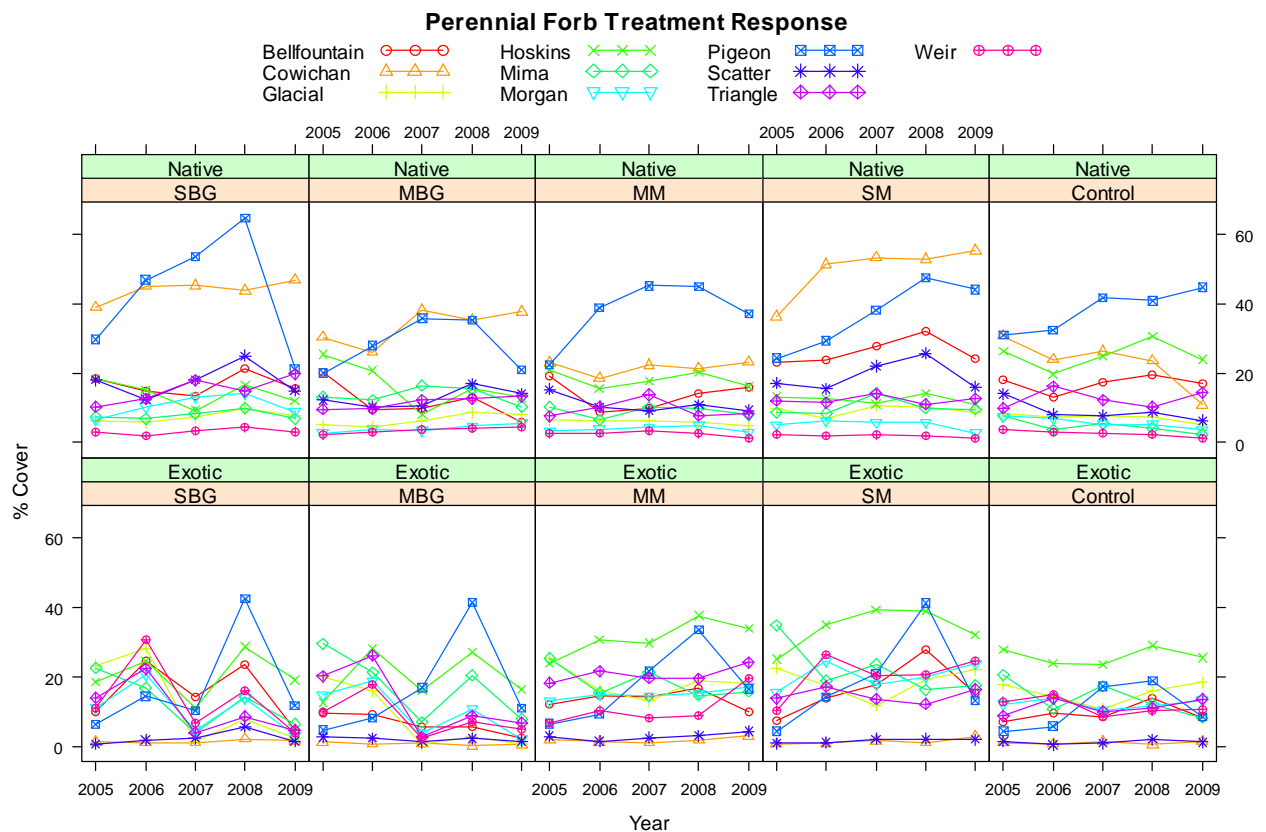




Figure 5. Response of perennial forbs to treatments at each site.

Although to date we have examined the responses of only a few native species individually (Table 5), we have not found deleterious effects of the treatment combinations using herbicides on native forbs, with the exception of *Fragaria virginiana* described above. For example, *Camassia quamash* had neutral responses to the SM and MM treatments, and neutral to positive responses to SBG and MBG. Two species, *Viola adunca* and *Prunella vulgaris*, declined in the MM treatment; the early summer mow likely interfered with seed production.

¹ P <0.05 using post-hoc tests following PERMANOVA

² P <0.05 using post-hoc tests following PERMANOVA

Table 5. Response of key species to management treatments. ↓↓ = large decline; ↓ = moderate declines; 0 = no response; ↑ = moderate increase; ↑↑ = large increase. If more than one response is listed, it indicates that the species responded differently at different sites.

Species	Sites	SBG	MBG	MM	SM
Invasive grasses					
<i>Agrostis spp.</i>	BF, GH, FH, MP, SC, TP, SW	↓↓	↓	0/↓	↓↓
<i>Anthoxanthum odoratum</i>	GH, CP	↓↓	0/↑	0	↓↓
<i>Arrhenatherum elatius</i>	PB, MM, SC	↓↓	↓↓	↓	↓↓
<i>Dactylis glomerata</i>	CP	↓↓	↓↓	0/↓	↓↓
<i>Holcus lanatus</i>	FH, PB	↓↓	↓	0/↓	↓↓
<i>Poa pratensis</i>	CP, MM, SC	↓↓	↓↓	0/↓	↓↓
<i>Annual bromes</i>	BF, CP, FH, PB	↓↓	↓	0	↓↓
<i>Vulpia bromoides</i>	FH	↑↑	↑	0	↑↑
Non-native forbs					
<i>Hypochaeris radicata</i>	BF, GH, FH, MM, MP, TP, SW	↓↓	↓↓	0	0
<i>Leucanthemum vulgare</i>	BF, GH, FH, MM, MP, TP, SW			0	↑
<i>Plantago lanceolata</i>	FH, MP, TP	0/↑	0/↑	0/↓	↓/↑
Native grasses, sedges, & rushes					
<i>Festuca roemerii</i>	SC, MM, TP	↓	↓	0/↓	0/↑
<i>Luzula comosa</i>	BF, GH, FH, MP, PB	↓↓	↓↓	0	↓
<i>Carex inops</i>	CP, GH, MP, SC, TP, SW	↑↑/0/↓	↑/0/↓	0/↑	↑↑/0
<i>Danthonia sp.</i>	BF, TP, MP, SC	↓↓	↓↓/0	0/↑	↓↓
Native Forbs					
<i>Camassia quamash</i>	CP, GH, MM, MP, SC, TP, SW	0/↑	↑↑/0	0	0
<i>Campanula rotundifolia</i>	MM	↑	↑	0	↑↑
<i>Fragaria virginiana</i>	BF, FH, PB, SC	low in 07/09 ²	low in 07/09 ²	0/↓	↑
<i>Lotus micranthus</i>	BF, GH, FH, PB	↑	↑	↑	0/↑
<i>Prunella vulgare</i>	MM	0	0	↓↓	0
<i>Microseris laciniata</i>	SC	up/down ³	↑	0	up/down ³
<i>Viola adunca</i>	SC	0	0	↓↓	0

¹*L. vulgare* initially increased, then declined sharply following burning + glyphosate; the pattern repeated.

²*F. virginiana* was low in the spring following burning + gly., but rebounded to pre-burn levels in 2008.

³*M. laciniata* had a large increase from 2005–2007, then declined from 2008–2009 to 2005 levels.

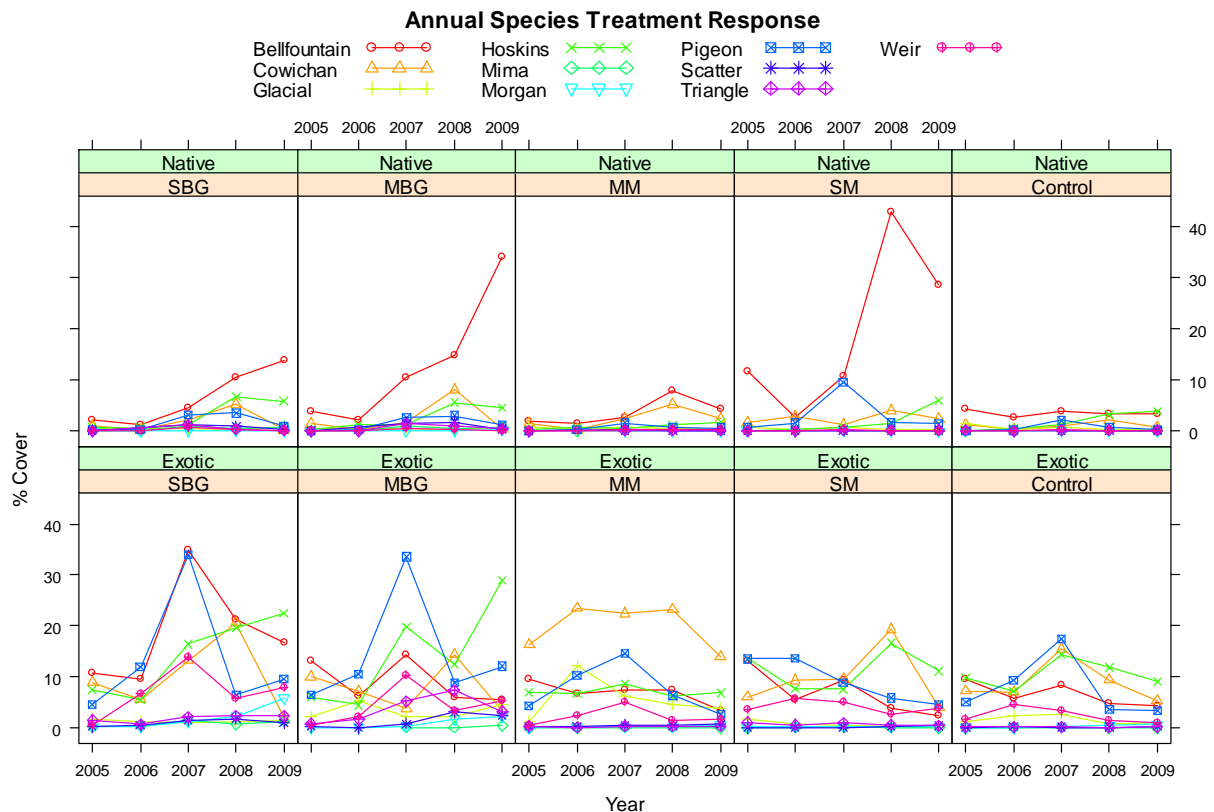


Figure 6. Response of annuals (grasses and forbs combined) to treatments at each site.

Non-native annuals

Non-native annuals increased dramatically at some sites after burning in both the SBG and MBG treatments (Figure 6). All sites in the Willamette Valley as well as Weir in Washington had this problem. All of these sites had a high initial diversity of non-native species, and South Weir and Fort Hoskins in particular had a low abundance and diversity of native species. Elsewhere in this ecoregion, similar post-fire increases in both native and exotic annual species persisted for 3-4 years, followed by resumed dominance of perennials (Dunwiddie 2002). These responses suggest that native annuals may be largely missing from many of these systems, and may need to be included in future native seeding applications to help fill the niche that was taken up by exotic annuals when exotic perennial grasses are controlled. Many sites had very little response by non-native annuals, presumably because these species were not present in the seed bank. At some sites repeated mowing led to an increase in non-native



Figure 7. *Plectritis congesta* in SBG at Mima Mounds.

annuals, but this response was usually small compared to the increase after burning.

Both annual grasses and annual forbs responded positively to burning. At some sites annual non-native grasses were abundant, but their distribution was patchy. Fort Hoskins had large increases in *Vulpia bromoides* in SBG plots, likely due to the creation of open space and the resistance of this species to sethoxydim.

Native annuals

Native annuals showed a similar pattern to non-native annuals, with the exception that no native annual grasses were found and native annuals were usually less abundant and diverse than non-native annuals. Native annuals such as *Madia gracilis*, *Lotus micranthus*, *L. purshianus*, *Triphysaria pusilla* and *Microsteris laciniata* all responded positively to burning, but not to other treatments.

Effects of treatments on ground cover

Degraded prairie landscapes are often characterized by a build up of thatch from invasive pasture grasses and a lack of bare soil. Fire suppression also leads to a thick layer of moss at many sites. The thick layers of moss and thatch likely inhibit plant establishment by seed. We found that burning was most effective at reducing moss and thatch and increasing bare soil (Figure 8)¹. Treatments that included repeated mowing (SM, MM) decreased thatch to some extent (Figure 8) even though material was not removed from the plots. This is likely because mowing chopped the material into smaller pieces, changed much of it from standing dead to material in contact with the ground, and thus speeded decomposition. However, the mowing treatments (SM, MM) did not decrease moss or increase bare soil (Figure 8).

Research question (3): Does native plant diversity and abundance increase rapidly in response to the reduction of the dominant grasses, or is seed addition required?

We hoped to see that the abundance and diversity of native plants would increase following the removal of invasive grasses. While we were pleased that our intensive treatment combinations had little deleterious non-target effects on native plants, at most sites we did not see a substantial increase in native abundance. The sites where the cover of native vegetation increased substantially (Cowichan, Pigeon Butte, Scatter Creek) were the sites that started with the highest initial abundance and

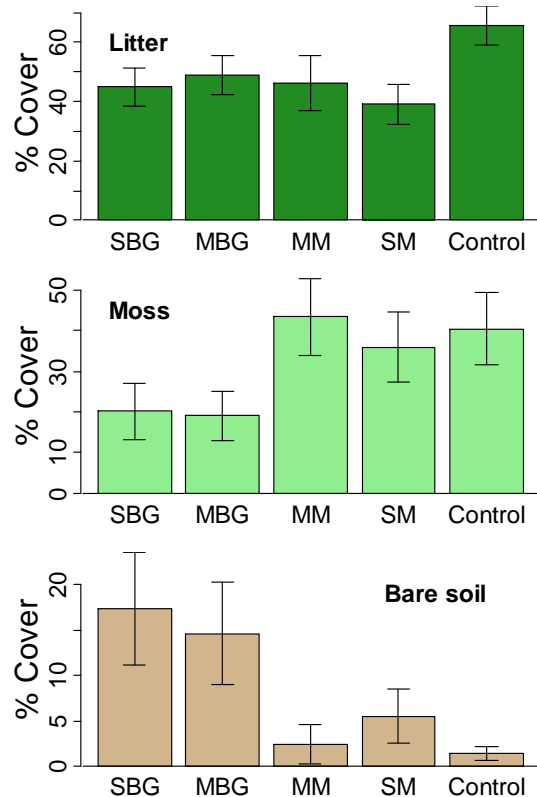


Figure 8. Treatment effects on ground cover in 2009

¹ SBG & MBG had significantly higher cover of bare soil (Tukey tests; linear mixed effects model)

diversity of native species. Sites that were highly degraded, like Fort Hoskins, showed little native response after native grass removal; there were simply few native plants to respond. One possibility is that native plants responded to invasive grass removal in other ways not detectable by tracking changes in cover, such as increased flower production or root growth. We did not see an increase in native diversity in any of the treatments in the absence of seed addition (Figure 10).

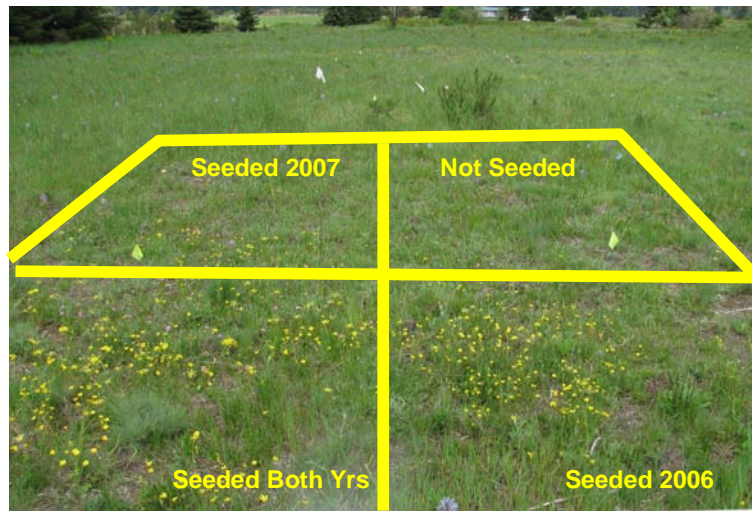


Figure 9. SBG plot at Scatter Creek, showing seeded species *Lomatium utriculatum* and *Ranunculus occidentalis* flowering.

This indicates that increasing native diversity requires seed addition, as the seed sources and seed bank are insufficient to allow natives to recolonize areas after invasive control. We found that seeding just 7 common species increased the diversity at the plot scale by an average of 3.4 species, and that this increase in diversity occurred regardless of the treatments. We also found the cover of those seeded species was highest in the burned treatments. While the increase in cover from seeding was not large enough to significantly affect the values for total native cover by the end of our study, over time we anticipate this will change as these plants continue to grow.

Seeding effects on native diversity

Seed addition was very successful at increasing native diversity (here defined as richness, the number of species). Seeding increased total native richness by an average of 3.4 species per plot across all treatments. What we found interesting is that seeded species established in all plots, even controls. Disturbance treatments alone had no effect on native richness (Figure 10); seeding was required, and successful, for increasing native diversity. With the exception of *Plectritis congesta*, the species we added are commonly occurring throughout the ecoregion, so it was surprising to us how much this seeding increased diversity at the plot scale. At the scale of the site or reserve, while many of these species may be present, they are sufficiently rare to only occur infrequently in the plots prior to seeding (Figure 13).

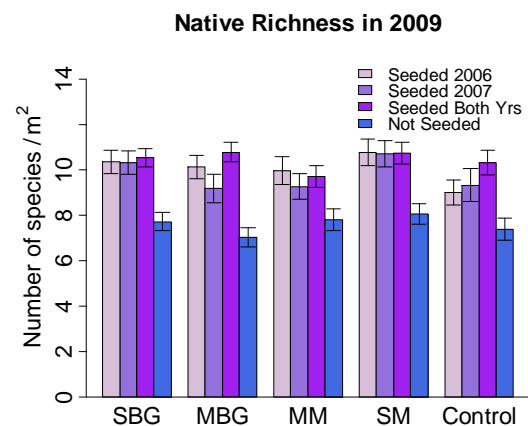


Figure 10. Seeding effects on native richness across all sites.

Seeding effects on native cover

While seeded species established in all treatments, seeded species achieved the highest cover in the 2 burn treatments, SBG and MBG (Figure 14). While at least some seeded species established at all sites and in all treatments, seed addition did not have a significant impact on total native cover. With the exception of the annual *Plectritis congesta*, most seeded species were slow to reach significant size and thus did not substantially change total cover values. By 2009 several of the perennial species that had been seeded in 2006 were flowering, particularly *Lomatium utriculatum* and *Ranunculus occidentalis* (Figure 9). The annual *Plectritis congesta* persisted in the seeded subplots, successfully reseeding itself. It did not spread substantially beyond the seeded area, indicating short seed dispersal distances.

When examining the cover of just the seeded species (Figure 12), we see many of these sites had some of these species present prior to seeding. After seeding, the cover increased substantially in the SBG and MBG treatments. The combination of seeding and the SBG treatment doubled the cover of seeded species compared to unseeded controls (4% cover to 8% cover). Both seeding in 2006 and seeding in both years showed the most increase in cover. The 2006 seeding provided the greatest benefit in SBG and MBG and the second seeding, in 2007, provided minimal additional gain (Figure 12). These data suggest that seeding into sites immediately after burning is much more effective than seeding into unburned areas or delaying seeding into burned habitat.

Factors influencing seeding success

Seeding after burning appears to be more effective than seeding even just 1 yr after a burn. Seeding without burning provided only minimal gains; while seeding increased diversity even in control plots, seedlings in controls were small, rare, and did not affect cover values (Figure 14; Karen Reagan, pers. comm.). Reducing invasive grasses alone (treatment SM) was insufficient to promote seedling establishment. Treatments which included burning were the only ones to significantly reduce thatch, moss, and increase bare soil.

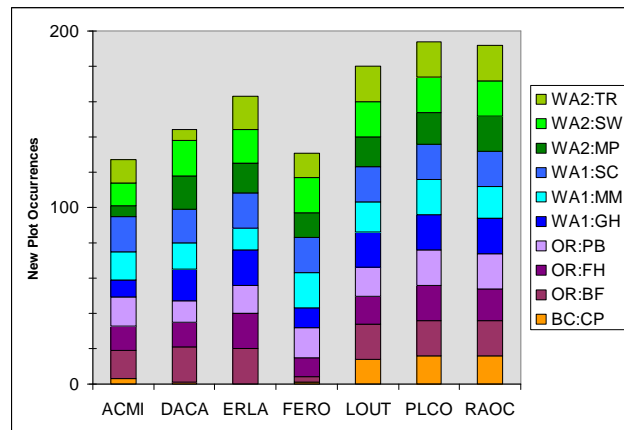


Figure 11. The number of new plot occurrences in 2007 for each seeded species at each site (total possible of 20 plots x 10 sites = 200). For species with less than 20 occurrences at a site, either the species was already present or it did not establish.

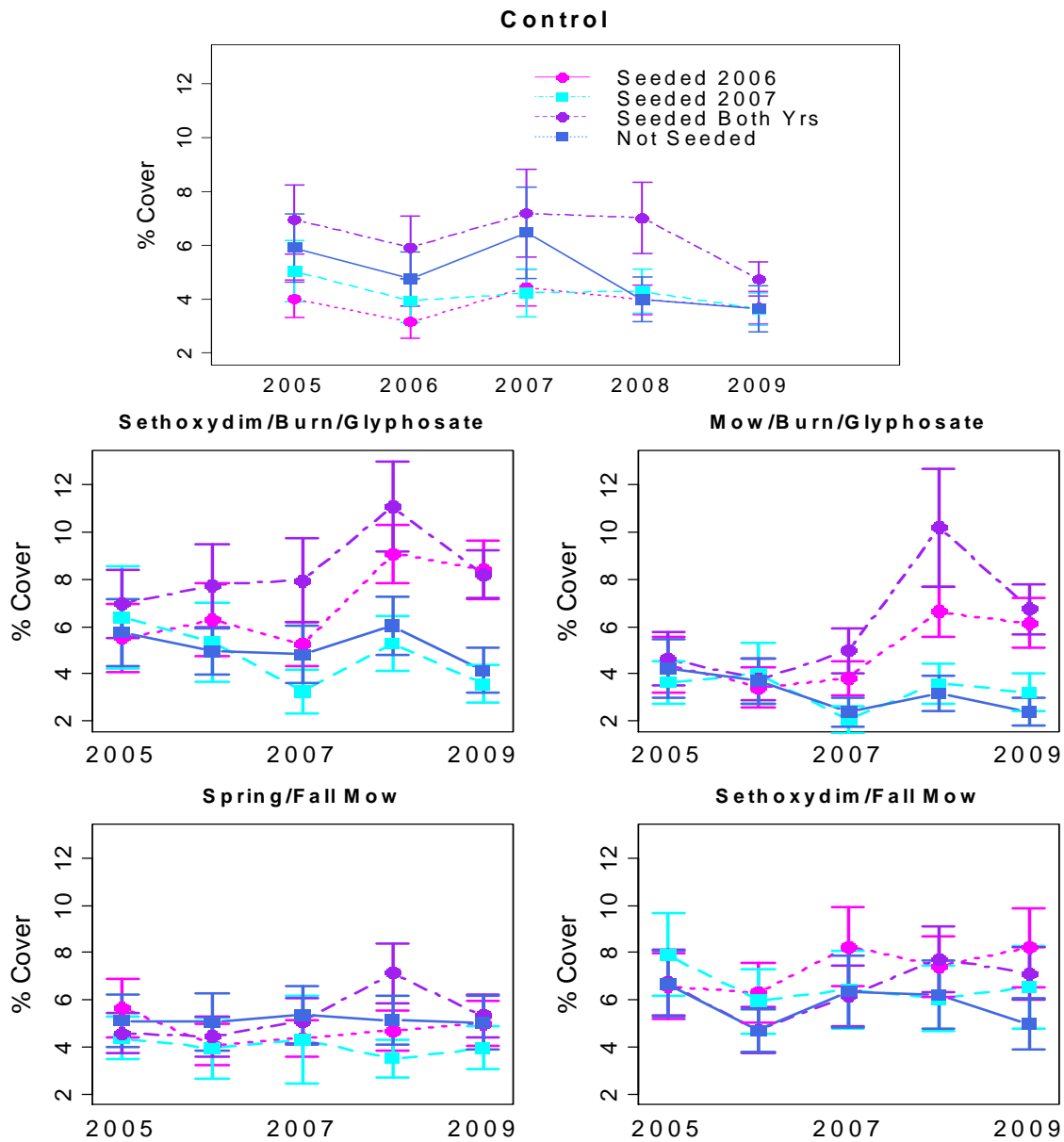


Figure 12. Total cover of all seeded species across all sites with treatments and seeding.

At many sites, we saw that the combination of invasive grass removal and burning (SBG) led to the highest cover of seeded species (data not shown). Some sites had very low cover of seeded species even in the SBG treatment. Very dry, shallow soil sites (Glacial Heritage, Morgan Property, South Weir) had low rates of recruitment (Karen Reagan, pers. comm.). The combination of high moss cover and low soil moisture may have reduced seedling success.

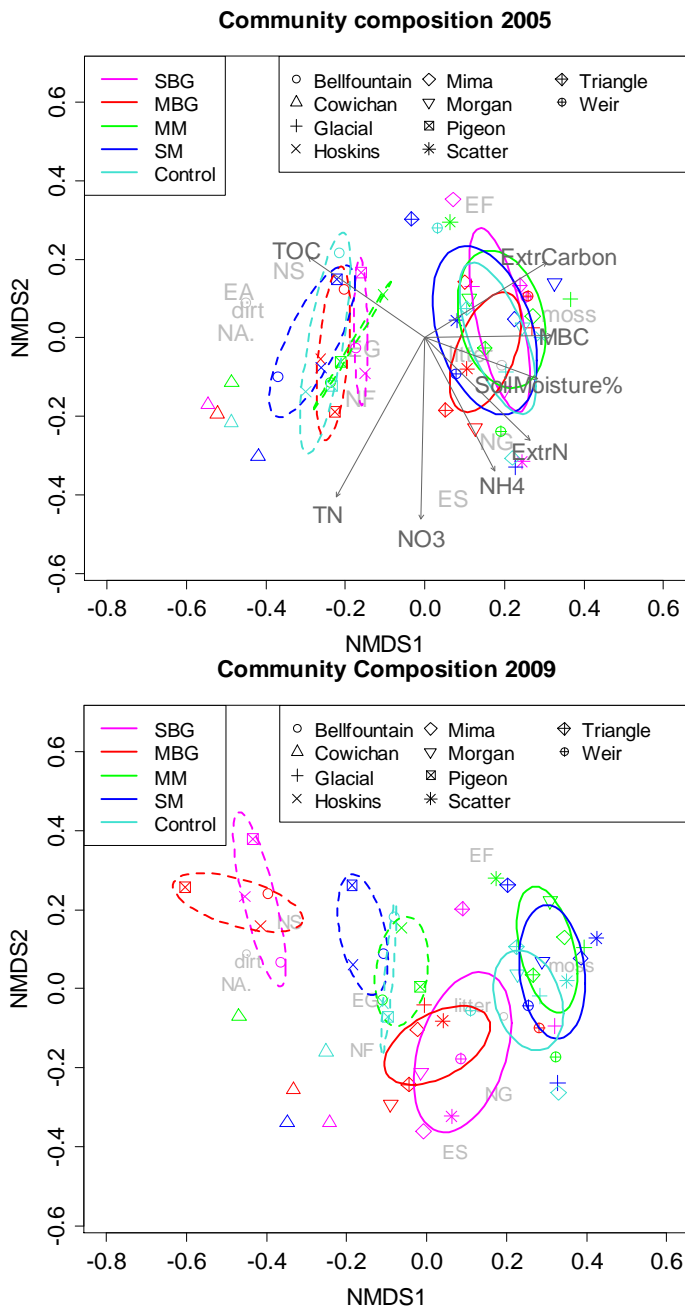


Figure 13. NMS ordination of treatments at each site in 2005 (top panel) and 2009 (bottom panel). Both years were ordinated simultaneously, but shown separately for clarity. Each point represents the centroid of the 4 replicates for each treatment. Ellipses show the standard error of the mean for Washington sites (solid lines) or Oregon sites (dashed lines). Environmental variables from the 2005 soil sampling are overlaid on the top panel; significant variables are TN=total nitrogen; NO3 = nitrate; NH4 = ammonium; ExtrN=total extractable nitrogen; TOC = total organic carbon; MBC= microbial biomass; ExtrCarbon = total extractable carbon.

Research question (4): Are results consistent across the ecoregion, or do treatment responses vary between sites?

While some results were consistent across the ecoregion, others varied widely between sites, likely depending on both the initial community composition as well as abiotic variables. At all sites we saw that sethoxydim was highly effective at reducing non-native grasses (Figure 4), and that at most sites non-native forbs were first to capitalize on this reduction (Figure 5). The response of the existing native plant community was more mixed, and sites with higher initial site quality (lower exotic abundance and diversity, higher native abundance and diversity) had the strongest response to treatments (Stanley et al. in prep).

While all sites saw an increase in the native species richness with seeding, the abundance of those species varied between sites (data not shown). Investigations of the response of seeded species following removal of non-native grasses showed an interesting pattern across sites (Richardson et al. in prep). We first created a 'site stress index', which ranked sites according to how stressful they are for plants – nutrient availability, soil moisture, rainfall, etc. We found that at low stress sites, the treatments that eliminated non-native grasses had the highest seedling recruitment. At high stress sites, however, the treatments with the highest non-native grass cover had the highest

seedling recruitment. This indicates that facilitation, in which non-native grasses somehow enhance the survival of new seedlings, may play a role in higher stress sites (Richardson et al. in prep). For example, non-native grasses could actually increase seedling survival through shading at sites with low soil moisture. However, non-native grasses are detrimental to the long-term success of native species, so finding new ways to increase seedling establishment at high-stress sites (e.g., through mulching, hydroseeding, establishing native grasses first, etc.) should be a management and research priority.

We can see that the 10 sites in our study varied in species composition at the start of the study (**Error! Reference source not found.**, top panel). We used NMS to ordinate sites based on functional groups (since there was high turnover of species across sites). The Washington sites all clustered together, as did the Oregon sites, with the Cowichan site in British Columbia on its own. It was interesting to us that the major differences in soil types across the ecoregion were also reflected in a difference in functional group assemblages at these sites. Sites in Oregon had lower microbial biomass, soil moisture, and soil nitrogen, but higher total organic carbon. The Oregon sites were also characterized by more bare soil and annual species, while the Washington sites had more perennials and moss cover. At the end of the study in 2009 (**Error! Reference source not found.**, bottom panel), we see a large divergence between treatments. Interestingly, functional group composition changed most dramatically in the two burn treatments; the other treatments were not much different from controls. The differing patterns we see between sites in their response to treatments may be due to the initial differences in functional group composition (**Error! Reference source not found.**, top panel).

Research question (5): Are results from small-scale experimental plots consistent with results at larger scales?

Results from the large, unreplicated plots (see methods page 8 and Table 4, page 8) were mixed, and interpreting the results from the large plots is challenging as the treatments were not applied as consistently. Particularly at the beginning of these studies, we needed to collect pre-treatment data prior to herbicide application, which often delayed the first herbicide application past the optimum phenological stage of the target grasses. Other issues arose with access, staff resources, and obtaining permissions for burning. Even so, the large plots did show some similar patterns to the small replicated experimental plots. At Bellfountain, Mima, and Weir, we saw large declines in exotic grasses (**Error! Reference source not found.**), with more moderate declines at the other sites. Treatments at Glacial were the least successful. The site was dominated by *A. elatius*, which we found to need several applications of Poast to achieve good control in the small plots. Although the Glacial plot was sprayed twice, the first spray had poor success as it was sprayed too late in the year to accommodate pre-treatment sampling. Additionally, the dense thatch layer, which was still present since the plot was not burned, may have inhibited good chemical contact.

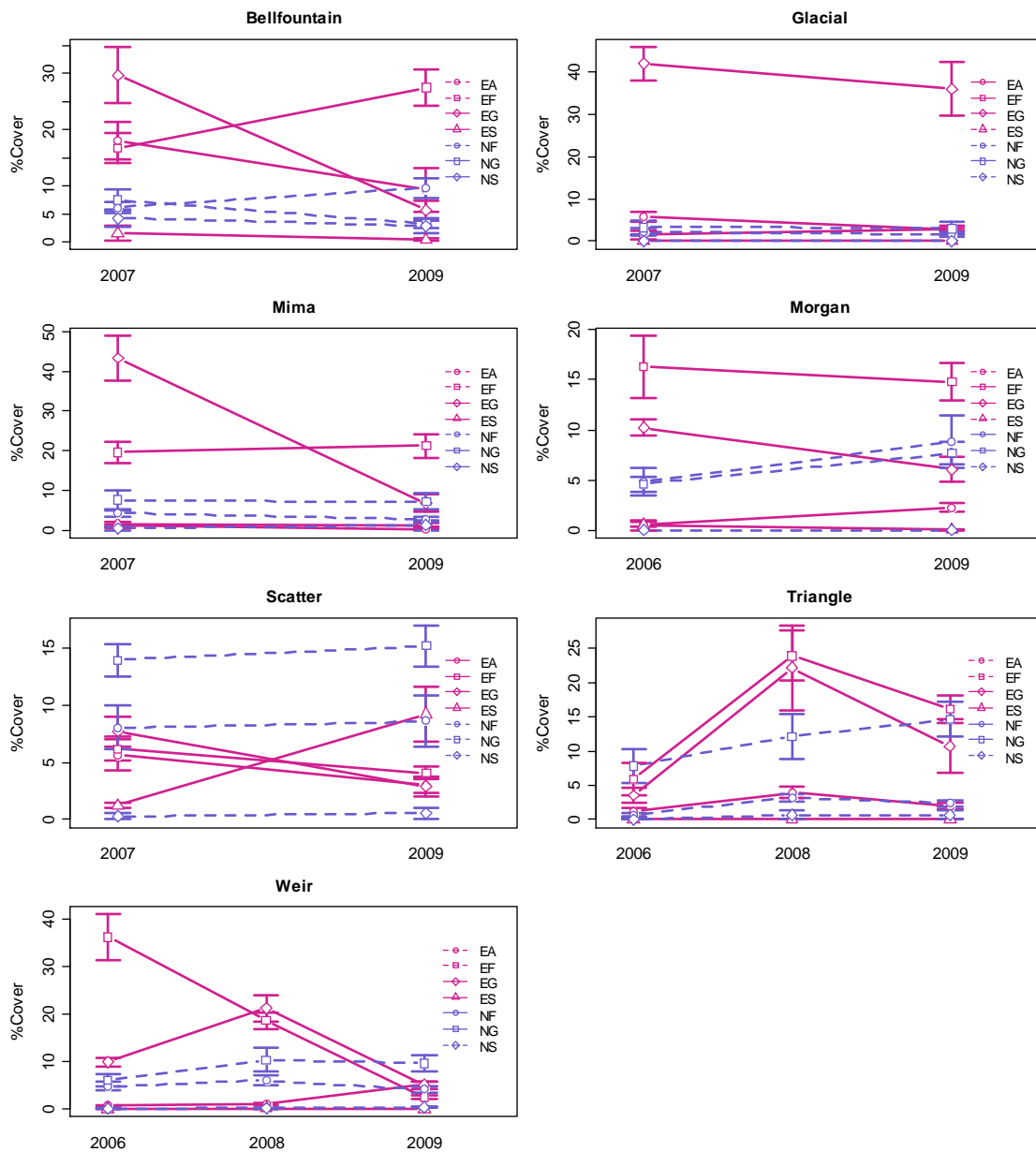


Figure 14. Functional group response to treatments in large scale plots. See Table 4 on page 8 for details of treatments applied to each large plot. (EA = exotic annuals; EF = exotic perennial forbs; EG = exotic perennial grasses; ES = exotic shrubs; N.an = Native annuals; NF = native perennial forbs; NG = native grasses, sedges, and rushes; NS = native shrubs.)

We see variation in which other groups responded to a decline in the exotic grasses. Exotic forbs increased most at Bellfountain, which had only received Poast applications, not burning + glyphosate, at the time of sampling in 2009. This site was burned, sprayed, and seeded in fall 2009 and visual observation in 2010 showed a decrease in adult *Leucanthemum vulgare*, but with many seedlings. Many of the seeded species did establish. We hope to monitor this site in 2011 to quantify its progress. At Scatter creek, exotic shrubs, almost entirely *Cytisus scoparius*, increased

over the study period. The treatment area at Scatter Creek was surrounded by mature *Cytisus* and had a large seed bank. At Morgan, both native forbs and native grasses increased with treatments (*F. roemerii* was seeded in this plot). At Weir, which had a successful implementation of the SBG treatment (however with the replacement of Fusilade, fluazifop, instead of sethoxydim) we observed a pattern similar to the small scale SBG treatments. As exotic grasses declined, exotic forbs increased sharply, and then declined following the burn + glyphosate treatment in fall 2008. Triangle was problematic in that quack grass (*Elymus repens*) increased dramatically in the plot from 2006 – 2008. Additionally, the planned sethoxydim spray in 2007 could not be carried out due to access issues. While the Fusilade applications in 2008 and 2009 reduced the *E. repens* somewhat, this grass proved remarkably tough to eradicate. With the reduction in *E. repens* we see an increase in native grasses, particularly *F. roemerii* and *E. glaucus*.

The large scale plots confirm results from the small – scale experiments, that grass-specific herbicides can be very effective, and burning + glyphosate has additional benefits in controlling both exotic grasses and forbs. However, the large plots highlight the difficulties of implementing these treatments at scale, even with a dedicated group of land managers. Obtaining permissions, gaining access, finding personnel time to carry out treatments, and optimizing the timing was much easier at the small scale. These data show that these treatments can be successful at the large scale as well, but that careful attention need to be paid to the logistics and timing.

MANAGEMENT RECOMMENDATIONS

Our experiments show very promising results for improving management strategies for restoring degraded prairies. We ranked the treatments based on how well they accomplished several objectives: 1) reducing non-native perennial grasses; 2) reducing non-native perennial forbs; 3) preventing increases in non-native annuals; 4) minimizing non-target effects on native species; 5) increasing abundance of existing native vegetation; 6) reducing moss & thatch and increasing bare soil; and 7) allowing successful seedling recruitment of seeded native species (Table 6). We created a decision tree based on these results (**Error! Reference source not found.**).

Controlling weeds and minimizing non-target effects

We found herbicides, and combinations of herbicides and burning, were very effective at controlling perennial herbaceous weeds while having minimal non-target effects. We were able to use relatively intensive management activities that achieved substantial gains with little negative impacts on the native plant community. As expected, susceptible native species were lost when using grass-specific herbicide (*Danthonia* species, *Bromus carinatus*, *Elymus glaucus*), but non-susceptible species were either unchanged or increased (*F. roemerii*, *Carex* species, *Luzula* species). We did not observe non-target effects of the grass-specific herbicide on any native forbs. Mowing achieved some reductions of perennial grasses at some sites, but these reductions were likely short-term reductions in stature. Mowing also stimulated expansion of non-native perennial forbs, and other ecological objectives were not met.

Table 6. Summary of treatment results for key ecological goals.

<i>How well did each treatment:</i>	Sethoxydim/ Burn +Glyphosate	Mow/ Burn +Glyphosate	Mow/ Mow	Sethoxydim/ Mow
Reduce non-native perennial grasses?	Best	Good	Poor	Best
Reduce non-native perennial forbs?	Best	Best	No – increased	No - increased
Prevent increase in non-native annuals?	Annuals increased, but increase temporary; reduced annual Bromes	Annuals increased, but increase temporary	No increase	Minimal increase; reduced annual Bromes
Minimize non-target effects on native species?	Sethoxydim-susceptible native grasses lost; other natives neutral - positive	Best - but native fescue may be reduced short-term	OK – however potential for negative long term impacts to native forbs from reduced seed production	Sethoxydim-susceptible native grasses lost; other natives neutral - positive
Increase abundance of existing native plants?	OK, depending on site quality	OK, depending on site quality	No	OK, depending on site quality
Reduce moss?	Yes	Yes	No	No
Reduce thatch?	Best	Best	Good	Good
Increase bare soil?	Yes	Yes	No	No
Allow successful seedling recruitment?	Best	Very good	Poor	Poor

Increasing native diversity and abundance

Even relatively high quality sites benefited from seed addition, which increased both native cover and native diversity, indicating strong seed limitation. If possible, we recommend native seeding at all sites. However, only treatments that included burning were able to significantly reduce thatch and moss cover and increase bare soil. We found that bare soil was required for seeding to be most effective. If burning is not possible, mowing is not an effective substitute to improve seeding success because it does not increase bare soil. We recommend exploring other alternatives to burning, such as grazing or dethatching to increase bare soil for seedling establishment.

Costs of treatment combinations versus benefits

We estimated the total 4-year costs of the tested treatment combinations at larger scale based on other large-scale restorations in the Willamette Valley (Table 7). The most effective treatments (SBG, MBG, and SM) are all much cheaper than repeated fall and spring mowing. While burning can be problematic, when done at large scales can be very cost-effective, as it often costs the same to burn 1 acre or 30.

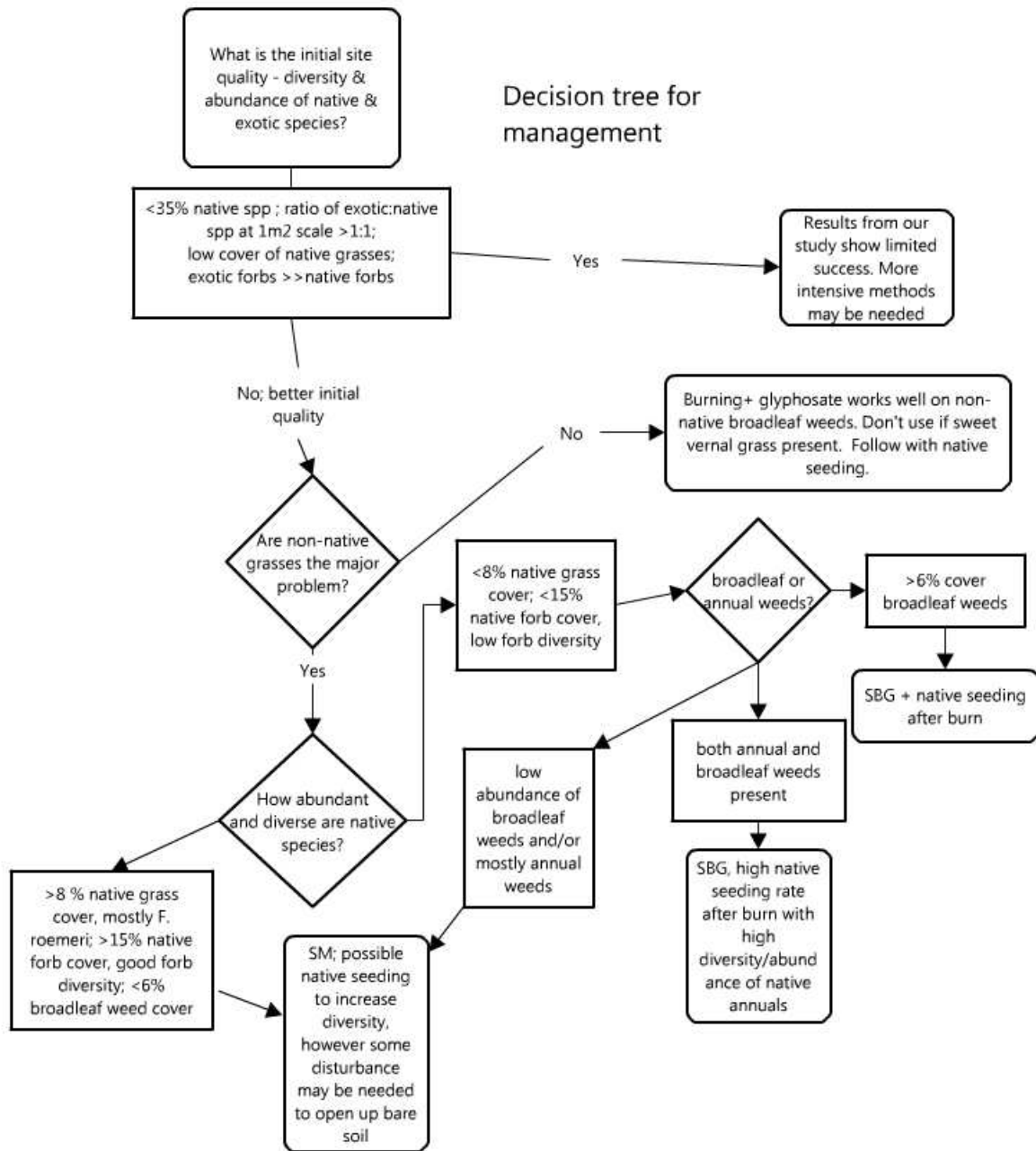


Figure 15. Decision tree for management of upland prairie sites based on initial vegetation and site conditions.

Which treatment combination is the most suitable for a particular site depends a good deal on the initial conditions and the ecological goals. We do not recommend repeat mowing generally, as it does not control herbaceous weeds, does not increase seeding success, and is costly compared to other treatments. We recommend that mowing be used instead to control shrubs where it has some demonstrated success. If other management options are not available, mowing can reduce tall-statured grasses

(particularly *Arrhenatherum elatius*), albeit temporarily. For this purpose, we recommend mowing in mid-season to cut off flowering heads (Wilson and Clark 2001), and that the mowing height be high enough to avoid most native species. The SM treatment works well at sites that retain good native diversity and abundance (so additional seeding is not planned), have a low abundance of exotic forbs, and the most pressing problem is one or more species of exotic perennial grasses. Recognize that susceptible native grasses will be lost, and may be difficult to seed back in without additional treatments to increase bare soil. The SBG treatment works well at sites with both exotic grasses and forbs. We recommend that seeding follow the burning if it all possible, and particularly where native diversity is low. The MBG treatment works nearly as well (as long as *Anthoxanthum odoratum* is not present), and has a cost savings over SBG. Our impression is that the initial mow in this treatment combination was unnecessary; the benefit derived from the repeat of the burning + glyphosate combination at a 2 year interval. One burn followed by glyphosate provided some gains, but the perennial grasses still had a significant presence and we saw a flush of non-native forb seedlings. The second burn + glyphosate reduced the perennial grasses substantially as well as the non-native forbs. If at all possible, with either SBG or MBG, we would recommend seeding native species after each burn.

Table 7. Estimated cost per acre for treatment combinations over 4 years, based on restoration costs in the Willamette Valley (burn \$35 - \$100/acre, with costs/acre decreasing with larger burns; mow \$50-\$177/acre, depending on distance, difficulty, shape, and thatch accumulation; broadcast spray \$25-\$105/acre depending on chemical costs, number of acres, distance, etc).

Treatment	Components	Low cost estimate, per acre	High cost estimate, per acre
SBG	3 broadcast Poast applications, 2 burns, 2 glyphosate applications	\$180	\$725
MBG	1 mow, 2 burns, 2 glyphosate applications	\$164	\$587
MM	8 mow	\$400	\$1,416
SM	3 Poast applications, 4 mow	\$266	\$1,027

CONCLUSIONS

Habitat restoration in Pacific Northwest prairies is impeded by abundant and widespread non-native plants (especially grasses), limited native plant reproduction, and landscape fragmentation. We used large-scale collaboration between scientists and managers to develop and test treatment combinations and advance restoration science and practice in our region. We address 5 key research questions in this report:

1. *Can combined treatments reduce the abundance of non-native grasses, and which combinations are most effective?* We found that treatment combinations including grass-specific herbicide were very effective at reducing the abundance of non-native grasses. Repeated burning followed by glyphosate reduced non-native grasses but not as much as sethoxydim. Mowing was ineffective.
2. *How do non-target species (native and non-native forbs, native grasses, and annuals) respond to treatment combinations?* Our most intensive treatment combinations had minimal harmful impacts to native species. For example, Roemer's fescue, a prominent native upland prairie grass, was resistant to grass-specific herbicide. Additionally the combination of burning followed by glyphosate effectively controlled re-sprouting weeds, reduced unwanted non-native grasses and forbs while generally retaining or increasing native abundance.
3. *Does native plant diversity and abundance increase rapidly in response to the reduction of the dominant grasses, or is seed addition required?* Native plant diversity and abundance did not increase rapidly in response to the reduction of dominant invasive grasses. Regrowth of natives was gradual and only seed addition increased native plant diversity. Native prairies in our region appear to be severely seed limited, and planting is crucial for increasing diversity and abundance.
4. *Are results consistent across the ecoregion, or do treatment responses vary between sites?* Some functional groups responded in predictable ways to treatments at all or most sites. For example, susceptible perennial grass species declined when sprayed with grass-specific herbicide. Non-native forbs often declined sharply with burning followed by glyphosate, and annuals tended to increase quickly after burning. While non-native annuals responded most, native annuals responded in a similar manner when present. Also, mosses typically declined after burning. Because of their consistency, these responses by some functional groups can be generalized to other sites. We also found considerable site-to-site variation in the magnitude and even direction of some treatment responses. For example, seedling establishment at low stress sites was lower where invasive grasses were abundant, but at high stress sites invasive grasses may have facilitated native seedling growth.
5. *Are results from small-scale experimental plots consistent with results at larger scales?* Results from 5 x 5 m plots appear to scale-up to restoration-level applications. The strongest evidence for this was that grass-specific herbicide reduced grass cover in large-scale treatments, particularly where application was well-timed and unhindered by thatch accumulation.

Managers must often struggle to balance restoration goals with the limited resources at hand. It is often hoped that weed control is sufficient for degraded landscapes that retain native species. While native species are constrained by competition with non-native dominants, we found that weed control alone does not lead to increased native diversity. In other words, the native seed bank is completely lacking at most of the existing prairie remnants, and native species establishment is constrained both by dispersal limitation (reaching a new site) and propagule limitation (generating enough seed compared to non-native species to colonize available space). Sites that lack a diverse native plant community were more likely to see the emergence of new non-native dominants following weed control efforts. Increasing native diversity is essential to the long-term success of prairie restoration in the Pacific Northwest. This is of course problematic as native seed is very costly, if it is even available.

Large-scale collaboration between scientists and managers can result in innovative treatment combinations backed by experimental rigor for rapid advances in restoration science and practice. This project could not have been completed without dedication and hard work from all our partners and collaborators. We hope these results will be valuable to restoration practitioners and restoration ecologists throughout the ecoregion.

PUBLICATIONS FROM THIS STUDY BY IAE

Stanley, Amanda G., Thomas N. Kaye, Peter Dunwiddie. 2008. Regional strategies for restoring invaded prairies: a multi-site, collaborative approach for controlling invasive species. *Native Plants Journal* 9(3): 247-254.

Stanley, Amanda G., Thomas N. Kaye, Peter Dunwiddie. In Press. Multiple treatment combinations and seed addition increase abundance and diversity of native plants in Pacific Northwest prairies. *Ecological Restoration*.

Bidilspach, Conor, Christine Williams, and Carl Elliot. 2009. Upland Prairie Viola Gets a New Lease on Life. *Bulletin of the Native Plant Society of Oregon* 42(2):13-21.

ADDITIONAL RESEARCH BY COLLABORATORS

Strong seed limitation in prairie grasslands of the Garry oak ecosystem

Reagan, Karen, P. Dunwiddie, T. Kaye, A. Stanley. In Preparation. Strong seed limitation in prairie grasslands of the Garry oak ecosystem.

Contact: Karen Reagan University of Washington Biology, Box 351800, Seattle, WA 98195-1800; sphitz@u.washington.edu

Abstract: How best to allocate limited resources is a critical issue in many conservation and land management decisions. In Pacific Northwest prairies, land managers must often decide whether it is necessary to sow costly native seed to enhance the diversity of native grasslands. Alternatively, will native species recover and repopulate sites once competitors are removed? To address these questions, we conducted a five-year study across nine field sites in two states to explore better ways to control invasives and enhance native diversity. At each site we established arrays of 20 plots – five treatments with four replicates. We tested four combinations of multi-year treatments directed at controlling invasives, including burning, herbicide, and mowing. The fifth set of plots was left untreated, and serving as a reference. Using a split-plot design, we used four levels of seed addition (none, seeding in 2006, seeding in 2007 and seeding in both years) to examine the significance of seed limitation, the treatment effect, and the interaction between the two. Pooling data across all sites, the number of native seedlings was significantly greater when seed was added suggesting that seed limitation strongly constrains the abundance and diversity of native plants in these prairies. Pre-seeding site preparation was also significant, but to a lesser degree. Treatment effects varied among the added species, although treatments that included burning tended to increase the number of seedlings of all species.

Effects of invasive grasses on soil moisture and competition

Davy, Lesley Erin. 2008. The influence of competitive interactions on the soil moisture dynamics in an invaded Garry Oak savanna. MS Thesis, University of Guelph. Advisor: Dr. Ze'ev Gedalof.

Contact: Ze'ev Gedalof, zgedalof@uoguelph.ca, Associate Professor, Climate & Ecosystem Dynamics Research Lab, Department of Geography, University of Guelph, Guelph, ON, N1G 2W1, 519.824.4120 x58083, <http://www.uoguelph.ca/geography/faculty/gedalof.htm>

Abstract: In this thesis I investigate the soil moisture budget for invaded and uninvaded Garry oak (*Quercus garryana*) associated ecosystems, and describe differences in seasonal phenology of Garry oak seedlings and their exotic and native associates. Garry oak savannas, a complex of grassland and transitional forest, are especially sensitive to these biological invasions. In a split paired plot experiment at the Cowichan Garry Oak Preserve in Duncan, British Columbia I measured soil moisture on treated and control plots at three depths in the soil over the growth season. When exotic vegetation was present, the drying rate of the soil was faster than in plots with native vegetation alone. Individual native and exotic species were also grown in a growth chamber and the seasonal phenology and soil moisture uptake were measured. When reproducing from seed, exotic vegetation remained metabolically active for longer than native vegetation – providing some insights into their competitive dominance in disturbed habitats.

Using plant traits to create predictive models for restoration

Clark, Deborah, M. Wilson, R. Roberts, P. Dunwiddie, T. Kaye, and A. Stanley. In Preparation. Plant Traits - a Predictive Tool for Habitat Restoration.

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Abstract: Most results of restoration efforts are species-specific and/or site-specific and therefore not general enough to be easily applied to other species and other sites. Our research addresses the issue of species-specific results by investigating the feasibility of using plant traits instead of taxonomic species to characterize species responses to restoration treatments. Location: Ten remnant bunchgrass prairie sites in the Pacific Northwest of North America (Oregon and Washington, USA; British Columbia, Canada). Methods: We compared two types of quantitative models for each of 10 prairie restoration sites: 1) plant trait models, which related plant traits to species field responses following restoration management treatments, and 2) species identity models, which related species taxonomic identity to species field responses following restoration management treatments. Species identity models determined the maximum amount of variability of field responses that can be explained by differences in individual species' responses to management treatments. Plant trait models determined what proportion of this explanatory power can be attributed to plant traits. Specifically, we used these two models to address the following questions: 1) How much of the variability in field responses of plants to restoration management treatments is explained by plant traits? 2) How well do plant traits explain the variability of field responses to restoration management treatments compared to models relating field responses to species identity? 3) Which of the two types of models – plant trait models or species identity models - is more parsimonious, i.e., a model that fits the data well with a relatively few number of explanatory variables? Results: 1) The plant trait models (relating plant traits with

plant field responses) explained much of the variability within each of the ten restoration sites, with R² values ranging between 31% and 69%, indicating significant predictive potential. 2) The species identity models (relating species taxonomic identity with plant field responses) explained between 47% and 74% of variability in field performance. Thus, the plant trait models explained nearly as much variability proportionally (0.60-0.93) as the species identity models, indicating that plant traits can potentially substitute for species. 3) In seven out of nine sites, the plant trait models were more parsimonious than the species identity models, indicating plant trait models are better than species identity models. Conclusion: Our results supports the feasibility of using plant traits as a common language to characterize field responses to restoration treatments, thus allowing results to be applied more generally, i.e., results are not limited to only the species included in the study. However, our results also indicate that our plant trait models are site-specific. We discuss the next steps in the development of more general and predictive models: incorporating environmental factors into the plant trait models to address the issue of site-specificity and testing the power of these models to predict vegetation responses.

Influence of exotic dominants on seedling establishment along a latitudinal stress gradient

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Abstract: The net outcome of species interactions – positive vs. negative - is contingent upon environment. With global environmental change this relationship will be fundamentally altered if negative interactions become prevalent, as is often assumed. There have been no large-scale experiments examining the persistence of positive interactions in degraded systems, nor the impact of such changes on local diversity. Here, we demonstrate that establishment, growth, and persistence of savanna plant species in stressful environments increased significantly only when near-monocultures of exotic perennial grasses were present. In a ten site meta-experiment conducted for five years along a 500-km latitudinal stress gradient, negative interactions from these grasses were only detectable in low-stress environments, characterized by high rainfall and fertility. Our evidence for the stress-dependent impacts of dominant species, regardless of their biogeographic origin, has vital implications for understanding how communities may be comprised of independent versus interdependent species, and their likely responses to management.

Effects of treatments on seed rain

Jepson, R. M. 2007. Evaluation of different management treatments in invaded prairie restoration on Pigeon Butte, Willamette Valley, OR. Senior research thesis, Whitman College. Research Advisor: Dr. Amanda Stanley, IAE. Biology Thesis Advisor, Dr. Heidi E. M. Dobson.

Contact: Amanda Stanley, amanda@appliedeco.org

Abstract: Introduced species threaten the fragmented native grassland in Oregon by out-competing native species. To determine ways to limit this threat and restore these communities, different combinations of mowing and herbicide treatments were evaluated in Willamette Valley grasslands during the summer of 2006. Seed production of forbs and grasses was measured in treated areas using seed rain to evaluate the impact of four different treatments on non-native plant success. Seeds were collected every two weeks for fourteen weeks using seed traps of which there were four per treatment. Seeds were then counted and identified to species. Of the 80 plant species present at the experimental site, 31 were represented in the seed samples including seventeen non-native, six native, and eight unidentified species. All treatments including the control (left intact) had a dominance of non-native seeds. Two seasons of spring Poast herbicide application (herbicide) led to the greatest increase in native seeds, whereas three seasons of fall and spring mowing (Mowing II) led to the greatest decrease in non-native seeds. Overall in terms of restoration because of the risks of using herbicides, Mowing II seems to be the best treatment.

Effects of treatments and community composition on native plant establishment

Katherine D. Jones. Effects of community composition on establishment of rare plants and nectar sources for rare butterflies. Master's Thesis; expected graduation 2012.

Advisor: Tom Kaye, Institute For Applied Ecology,

Contact: Katie Jones, joneskat@science.oregonstate.edu, Department of Botany and Plant Pathology, 2083 Cordley Hall, Oregon State University, Corvallis, Or 97331. 509-499-1749

Abstract: An estimated 99.5% of the historic upland prairie ecosystem in Willamette Valley has been lost. Because the remaining patches of intact upland prairie have become biological islands, natural processes alone are no longer able to restore ecosystem processes and species. Active management, including vegetation manipulation and native species reintroduction, is necessary to restore and protect these habitats. I am exploring how site and seed preparation affect establishment of native perennial forbs by seed. My aim is to develop a set of scientifically tested tools to support land managers attempting restoration of upland prairie habitats and to enhance the likelihood that reintroduction of native plants will be successful.

This research focuses on five plant species: three endangered or threatened species—Kincaid's lupine (*Lupinus sulphureus* ssp. *kincaidii*), Willamette daisy (*Erigeron decumbens* var. *decumbens*), golden paintbrush (*Castilleja levisecta*); and two common plant species—rose checkermallow (*Sidalcea virgata*) and roughleaf iris (*Iris tenax*). Three of these species are used by the endangered Fender's blue butterfly (*Icaricia icarioides fenderi*). I have introduced seeds or transplants of the five native

plant species into plots with varying levels of invasive grass, litter and thatch levels to determine minimum and optimal conditions that support seedling establishment of all five species. Field experiments will investigate the relationship between existing plant community composition and seedling establishment as well as test the effectiveness of seed preparation on establishment.

If establishment by seed is not effective, the next tool available to managers is using transplants grown in a greenhouse to reintroduce species to restoration sites. It is therefore essential that reliable methods of breaking seed dormancy are available to support greenhouse efforts. Because dormancy breaking requirements for rose checkermallow and roughleaf iris are not yet established, I will conduct a series of laboratory experiments at the OSU seed laboratory to identify germination requirements for these species.

Specific objectives of this research that will help achieve this goal are:

- Identify the relationship between invasive grass, associated litter and thatch and seedling establishment of the species tested.
- Determine if scarification of seeds with physical dormancy prior to field seeding increases establishment in experimental plots.
- Develop germination protocols for rose checkermallow and roughleaf iris.

Understanding the underlying science of habitat restoration in upland prairie ecosystems is fundamental to successfully preserving upland prairie species and the upland prairie ecosystem in the Willamette Valley.

Functional Groups, Traits, and the Performance of Species in Restoration

Rachael E. Roberts. 2007. Functional Groups, Traits, and the Performance of Species in Restoration. MS Thesis, Oregon State University. (Advisor, Deborah Clark, OSU).

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Abstract: In ecological restoration, species that are sown to increase the native plant diversity range in establishment ability. Some species readily establish, while others rarely do. This study set out to investigate some of the potential processes influencing species establishment, as well as the traits that are associated with the success of species in restoration. Twenty - eight species native to upland prairies of the Willamette Valley of Oregon were sown in different seed mixtures in field plots in a former agricultural field. These species were divided into three a priori functional groups, annual forbs, perennial forbs, and grasses, to determine whether interactions among functional groups influenced the performance of functional groups and other measures of restoration success, including native species richness, cover, and biomass. There was no evidence of inter - group competition; rather, competition was greater within functional groups, particularly within annual forbs. Native cover and biomass increased significantly with the number of functional groups sown; however, the amount of variation explained by functional group diversity was less than 10%. Non - native plant abundance was found to influence native performance much more than functional group richness. Sown native richness

was not strongly influenced by either functional group richness or non - native abundance. To look for correlations between species traits and performance, eleven different traits of each species were measured from both laboratory and field - grown plants. These were related to measures of field performance, including cover (%) and frequency of establishment using step - wise regression techniques. Models relating traits to measures of performance were strong, with traits explaining up to 56% of variation in cover, and 49% of establishment frequency. The relationship between traits and performance varied depending on functional group, and intergroup interactions among annual forbs also influenced cover within this functional group. If these results were to be put into practice, a functionally diverse seed mix for greater native abundance would be recommended for greater native cover. The regression models should be tested using different species or at a different site to determine their predictive ability. The results presented here should be useful to land managers and from a general ecological sense as well.

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