

Monitoring change in exotic plant abundance after fuel  
reduction/restoration treatments in ponderosa pine forests  
of Western Montana

by  
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Director: Carl Fiedler

Exotic species were monitored following treatments designed to reduce wildfire hazard and initiate restoration of forest structure and process in ponderosa pine (*Pinus ponderosa*)/Douglas-fir (*Pseudotsuga mensiezii*) forests to compare response among treatments. Treatments included: no treatment (control), prescribed burning, comprehensive cutting (thinning), and thinning followed by burning. Each treatment was replicated three times in a randomized block design. Frequency, richness, and cover of exotic species were sampled at multiple scales (1 m<sup>2</sup>, 100 m<sup>2</sup>, and 1000 m<sup>2</sup>) in 10 modified Whittaker plots per treatment unit. Data were collected before treatment, one growing season after burning, and two growing seasons after thinning. Special emphasis was given to a subset of harmful exotic species, which were considered undesirable.

Non-metric Multidimensional Scaling (NMS) revealed that several thin/burn plots were different in exotic species abundance and composition than plots in any other treatment. Blocked Multi-Response Permutation Procedures (BMRPP) detected significant treatment differences in exotic and undesirable species abundance at each scale. The thin/burn treatment consistently had the highest levels of overall undesirable and exotic abundance. Indicator Species Analysis revealed several exotic species became significant indicators of the thin/burn after treatment. Responses of individual exotic species differed depending on the scale sampled, with more significant differences in abundance being detected at larger scales. Regression models showed that exotic species richness increased with canopy openness, fire severity, and increased native richness, and decreased with sapling density, supporting a resource limitation model of exotic invasion.

## Table of Contents

	<u>Page number</u>
Abstract	ii
Table of contents	iii
List of tables	v
List of figures	vi
List of equations	vii
List of appendices	viii
Introduction	1
Literature Review	3
-Ponderosa pine restoration	3
-Exotic Invasion	5
Study site	15
Treatments	16
Field Methods	17
Summary Methods	21
Analytical Methods	23
-Non-metric Multidimensional Scaling	23
-Blocked Multi-Response Permutation Procedures	26
-Indicator Species Analysis	30
-Covariate Analysis	31
Results	34
-Exotic Plant Community Patterns	34

<b><u>Table of contents (cont.)</u></b>	<b><u>Page number</u></b>
-Scale	47
-Indicator Species Analysis	50
-Covariate Analysis	51
Discussion	58
-Differences among treatments	58
-Environmental variables	61
-Measuring exotic abundance	63
- Monitoring Individual Undesirable Species	65
- Other Notable Exotic Species	70
- Exotic impact	72
- Future monitoring and analyses	73
Conclusion	76
Acknowledgements	78
Appendices	79
Bibliography	80

## List of Tables

<u>Table</u>	<u>Page number</u>
1. Exotic species present at the study site	12
2. Site characteristics	15
3. Measured and calculated variables	24
4. Change in quadrat-level cover of exotics from pre-to post-treatment, and BMRPP tests of differences among treatments	40
5. Change in quadrat-level exotic frequency from pre-to post treatment, and BMRPP test for treatment differences	41
6. Change in plot-level frequency of exotics from pre-to post-treatment, and BMRPP tests for treatment differences	45
7. Subplot-level cover and frequency of undesirable species and BMRPP tests for treatment differences	48
8. Indicator Species Analysis before and after treatment application	52
9. Final regression model of change in plot-level exotic richness with block and treatment	53
10. Final regression model of change in plot-level exotic richness with environmental variables	54
11. Treatment differences for environmental variables significantly correlated with exotic richness change	57

## List of Figures

<u>Figure</u>	<u>Page number</u>
1. Treatment unit layout	18
2. Modified Whittaker plot layout	19
3. Number of axes verse stress for Non-metric Multidimensional Scaling (NMS).	36
4. NMS ordination of plots in exotic species space.	36
5. NMS ordination of exotic species in plot space.	37
6. Pre-treatment exotic richness at multiple scales	49
7. Post-treatment exotic richness at multiple scales	50
8. Missoula spring precipitation for 2000, 2001, 2003 and average precipitation.	64

## List of Equations

<u>Equation</u>	<u>Page number</u>
1. Change from pre-to post-treatment	22
2. Basal Area	22
3. Canopy completeness	22
4. Converted aspect	23
5. Effective aspect	23
6. Treatment effect	23
7. MRPP test statistic	27
8. Chance corrected within group agreement	28
9. Blocked MRPP delta	29

## List of Appendices

<u>Appendix</u>	<u>Page number</u>
1. Tree data before and after treatment	79
2. Burn variables	79

## **Introduction**

Fire exclusion and other anthropogenic activities since the arrival of Euro-American settlers in the inland west have led to structural and functional changes in fire-adapted ponderosa pine and pine/fir forests (Covington and Moore 1994; Allen et al. 2002; Fiedler et al. 1995). Conditions predominating in these forests today are thought to be outside the historical range of variability (Allen et al. 2002). Of principal concern has been the switch of fire regimes from frequent low intensity fires to stand replacing infrequent fires (Arno et al. 1995; Waltz et al. 2003; Touchan et al. 1996; Wright and Agee 2004). Many areas have high accumulations of fuel, increasing the likelihood of stand replacing wildfires (Covington and Moore 1994; Mutch 1970; Hessburg and Agee 2003; Fiedler et al. 2003). Restoration/fuel reduction treatments have been advocated as a method to return the structure and function of these ecosystems (Fiedler et al. 2000). These treatments may result in increased tree vigor (Feeney et al. 1998; Kolb et al. 1998; Cochran and Barret 1993), reduced risk of stand replacing fire (Pollet and Omi 2002; Fiedler et al. 2003; Fulé et al. 2002), and have the potential to make these systems self sustaining, given the historic disturbance regime (Fiedler et al. 1995; Arno et al. 1995).

Paradoxically, one problem that occurs after wildfire--invasion by exotic plant species--may also be facilitated by restoration treatments (Griffis et al. 2001; Crawford et al. 2001). Invasions pose a serious threat to ecosystems worldwide (Vitousek 1996; Pimental et al. 1999; Sieg et al. 2003). There is also considerable evidence that disturbance, such as that caused by restoration treatments, facilitate invasion (Petryna et al. 2002; Burke and Grime 1996; Hobbs 1991; Smith and Knapp 1999; Hobbs and

Huenneke). Despite the enormous body of literature concerning exotic invasions, the response after restoration has not been well studied (D'Antonio and Meyerson 2002).

The Fire and Fire Surrogate (FFS) study was funded by the USDA/USDI Joint Fire Sciences Program to evaluate restoration treatments in fire-adapted forests from a broad interdisciplinary perspective. The University of Montana's Lubrecht Experimental Forest, located in the Garnet Range of western Montana, is one of 13 sites in this nation-wide study. Four treatments were evaluated at Lubrecht: mechanical thinning, prescribed burning, mechanical thinning followed by prescribed burning, and an untreated control. These represent the basic options available for land managers (Fiedler et al. 2000). This study presents a unique opportunity to compare and contrast the effects of restoration treatments on exotic plant invasions in a randomized, replicated, block design in ponderosa pine/Douglas-fir forests of western Montana.

Specific objectives of this study include:

- 1) Compare the overall performance of exotic and undesirable species among treatments by comparing abundance among treatments.
- 2) If differences in overall exotic and undesirable abundance are observed, which species are driving these results, and to which treatment are they responding most strongly?
- 3) Determine environmental variables (whether related to treatment or not) that are associated with increased exotic richness.
- 4) Examine the effect of scale: does exotic species response to restoration treatments differ depending upon the scale of measurement?

## **Literature Review**

### **Ponderosa Pine Restoration**

There is considerable evidence of changes in ponderosa pine ecosystems in the past 100 years due to anthropogenic activities, especially fire exclusion. Historically stands were characterized by large fire resistant trees and vigorous understory plant communities, as frequent fires thinned the smaller trees (Stone et al. 1999; Fiedler et al. 1992; Mutch 1970; Arno 1988). Today many stands have been homogenized, consisting of dense tree regeneration, with stymied nutrient cycling, decreased herbaceous vegetative growth, and reduced tree vigor leading to large scale insect and disease outbreaks (Covington and Moore 1994; Fiedler et al. 1995; Allen et al. 2002). Large pines remaining from presettlement may be stressed due to intense competition from understory trees (Moore et al. 1999; Mast et al. 1999). Detrimental effects are not limited to the vegetation, as wildlife habitat, water relations, and species diversity can be impacted as well (Covington and Moore 1994).

The removal of fire has led to increased recruitment of shade tolerant species (usually firs) in place of seral species that evolved with fire and depended on it to maintain dominance (Fiedler et al. 1995; Covington and Moore 1994). Dense growth creates levels of fuel loading and continuity that predispose these forests to stand replacing wildfire (Covington and Moore 1994; Mutch 1970; Hessburg and Agee 2003). Indeed, the fire regime has been changed from frequent surface fires to large infrequent stand replacing fires that were formerly uncommon in these forests (Arno et al. 1995; Allen et al. 2002; Waltz et al. 2003; Touchan et al. 1996). However, without intervention the number of stand replacing wildfires may continue to increase. Eighty percent of the

pine/fir/mixed dry conifer forests in western Montana rate high or moderate for crown fire hazard (Fiedler et al. 2003). Detrimental effects of these fires may be long lasting. Heavy ponderosa pine seeds only disperse short distances, so landscape scars can remain long after a stand replacing fire, as trees may not reestablish for decades (Allen et al. 2002).

The prevalence of these conditions creates the need for restoration/fuel reduction treatments, which have been shown to decrease the severity of actual wildfires (Omi and Martinson 2002; Pollet and Omi 2002). In addition, treatments may increase the vigor of remaining trees, thereby increasing resistance against large scale disease and insect outbreaks (Feeney et al. 1998; Cochran and Barret 1993; Kolb et al. 1998). Restoration may be defined as “the process of reestablishing to the extent possible the structure, function, and integrity of indigenous ecosystems” (Society for Ecological Restoration 1993; Moore et al. 1999). In ponderosa pine forests this is accomplished by creating more open stand conditions, consistent with the historic range of variability that characterized these ecosystems in the past (Morgan et al. 1994).

Multiple treatment options are available for restoration. Fire was historically the main disturbance that shaped ponderosa pine ecosystems, regulating their structure and function (Moore et al. 1999). Reintroducing fire could align forests with climatic conditions, which were important in determining the fire regime in the past (Allen et al. 2002). Fire may stimulate understory vegetation, create seed beds, and transform nutrients into a more available form, especially nitrogen (Fiedler et al. 1995; Arno et al. 1995; Deluca and Zouhar 2000). In some cases fire alone has proven to be sufficient for ecosystem restoration (Allen et al. 2002). However, even though fire alone may be

sufficient to restore some sites, many others are so densely stocked that the probability of a stand replacing fire is too high, and fire would not function in the same role it did historically, and may even be detrimental (Moore et al. 1999; Swetnam et al. 1999; Mutch 1970).

Thinning or other silvicultural prescriptions could decrease the risk for stand replacing fire by decreasing fuel continuity (Fiedler et al. 2003), as well as restoring the forest structure and species composition (Fiedler et al. 2003; Allen et al. 2002). Thinning can stimulate herbaceous growth (McConnell and Smith 1970) and nutrient cycling (Kaye and Hart 1998) and offset treatment costs (Vissage and Miles 2003; Fiedler et al. 2003). However, it remains debatable whether thinning and other cutting treatments can substitute for fire (Swetnam and Baisan 1996; Arno et al. 1995). Combining thinning followed by burning could potentially restore both the structure and function of these ecosystems and provide the benefits associated with each treatment (Fiedler et al. 2000).

### Exotic Invasions

There is an expansive body of literature dealing with invasions by exotic plant species, but little documentation of the effects of restoration treatments on exotic plant invasions. Plant invasions are not a new phenomenon. They are a natural part of the earth's history and necessary for the persistence of life (Botkin 2001). When viewed at an evolutionary time scale plants are very mobile (Moore et al. 1999). Human activities in modern times have generated a breakdown in biogeographical barriers, which has greatly accelerated the rate of spread of exotic species (D'Antonio and Vitousek 1992). While the history of the earth has seen many plant invasions, the human transport of plants in the past 500 years dwarfs the movement of plant species during any other time

period of similar duration in the earth's history (Mack et al. 2000). Plant invasions are now common throughout the world with over 4000 species introduced to North America north of Mexico, which comprises 20 percent of the vascular plant flora in the region (Davis 2003).

Plant invaders threaten ecosystems worldwide, and may cause irreversible damage (Sieg et al. 2003). Although, exotic species are known to have drastic economic impacts (Pimental et al. 1999), the ecological consequences may dwarf the economic costs (Vitousek et al. 1996). The cause for concern is high because invasive species may alter plant communities (including relative abundances of plant species), as well as ecosystem properties and processes (Olson 1999). Invasions can have profound negative effects on biodiversity (Radosевич et al. 2003; Botkin 2001; Hettinger 2001). Indeed many endangered species today are threatened primarily by exotic species invasions (Czech and Krausman 1997). More species have gone extinct due to invasive species introductions than any other anthropogenic change, except for changes in land-use (D'Antonio and Vitousek 1992). Furthermore, invaders may not simply compete with native vegetation, but may fundamentally change the environment for years to come (Vitousek et al. 1996). See Blossey (1999) and Masters and Sheley (2001) for further details about the impacts of exotic plant invasions.

Plant invasions have generated interest for years (Elton 1958), but predicting what allows invaders to be successful has been elusive (Williamson 1996; Peterson and Vieglais 2001). A wide variety of environmental factors underlie the differential successes of invaders, including disturbance, resource availability, habitat fragmentation, evolutionary history, stress, community structure, propagule pressure, predation,

mutualism, and competition (Kolb et al. 2002; Alpert et al. 2000; Richardson et al. 2000; Davis et al. 2000). In a worldwide review of invasion, Lonsdale (1999) summarized the success of invasion as being determined by three broad categories; the traits of the species, the propagule pressure, and the invasability of the environment, which is an emergent ecosystem property. These factors interact with each other and the invading species in such a way that invasion success depends on the interaction of the species and the environment in which it is establishing (Alpert et al. 2000; Heger and Trepl 2003; Sher and Hyatt 1999).

Species richness is an additional, although perplexing factor that affects invasion success. Diverse assemblages have been hypothesized to utilize resources more fully, therefore more effectively excluding invaders (Levine and D'Antonio 1999; Elton 1958). However, diverse assemblages may also have additional resources and microhabitats that can support a larger number of exotic species (Levine and D'Antonio 1999). Increased richness has typically been shown to confer resistance to invasion at small scales in experimental studies (Tilman 1997; Levine 2000; Meiners et al. 2004; Naeem et al. 2000), but at larger scales in observational studies it has been correlated with increased invasion (Lonsdale 1999; Stohlgren 1999; Levine 2000; Smith and Knapp 1999). However, invasive species may be responding to environmental variables that co-vary with species richness and not species richness per se (Thompson et al. 2001; Levine 2000; Levine and D'Antonio 1999; Stohlgren et al. 2002).

Disturbance may also be a key factor in determining invasion success. Disturbance is a natural part of an ecosystem and may be necessary for maintenance of biodiversity (Huenneke et al. 1990). It can also facilitate exotic invasion, and may even

provide profound advantages to exotics (Lonsdale 1999). Numerous studies have documented the trend of increased invasion in disturbed areas (Hobbs and Humphries 1995; Hobbs and Huenneke 1992; Hobbs 1991; Lake and Leishman 2004). Disturbance may provide safe sites needed by exotics to establish (Masters and Sheley 2001; Hobbs 1991). Since many exotics are opportunistic species that can establish and gain dominance quickly, they may have advantages over natives after disturbance (Hessburg and Agee 2003). Disturbance has the potential to break down the physical barriers that keep exotic species out (Silveri et al. 2001), and reduces competition from native species (Alpert et al. 2000; Weaver et al. 2001; Lonsdale 1999). Disturbance is critical in the early stages of invasion because it creates openings that exotic invaders can exploit (Masters and Sheley 2001). Otherwise native species may be able to exclude the exotics from the system (Lonsdale 1999).

Resource availability is another key factor often tightly interwoven with disturbance. Adding resources has consistently been shown to promote invasion (Bakker and Berendse; Wedin and Tilman 1996; Thompson et al. 2001; Kolb et al. 2002; Claassen and Marler 1998; Huenneke et al. 1990). Increased levels of nutrients have also been shown to alter the competitive outcomes between native and exotic species-- favoring exotics (Blicker et al. 2002; Kolb et al. 2002). The fluctuating resource hypothesis proposed by Davis et al. (2000) aims to synthesize existing knowledge of what determines invasion success. Invasability may be increased whenever additional limiting resources are added to a system, or when resource uptake is decreased, both of which may occur after disturbance (Byers and Noonberg 2003; Davis et al. 2000; Huenneke et al. 1990). This also implies that invasability is not a static ecosystem property but varies

over time (Heger and Trepl 2003; Milbau et al. 2003). In support of this hypothesis, Burke and Grime (1996) found the highest levels of invasion when disturbance was combined with nutrient addition. Both exotic and native species may utilize resources similarly (Thompson et al. 2001; Meiners et al. 2004). This suggests that when resources are available, the first plant to get to them (whether native or exotic) has the potential to become established in the community (Davis 2003). However, resource addition may also shift the competitive outcome to favor exotic species (Kolb et al. 2002)

Thinning and burning have been documented as disturbance agents that increase exotic plant abundance in ecosystems throughout the world. Numerous studies indicate that fire may facilitate invasion by exotic species (Johnson 1998; Petryna et al. 2002; Hobbs and Huenneke 1992; Crawford et al. 2001). Burned areas have open ground, low competition, and high nutrient levels that favor colonization by invasive plants (Goodwin and Sheley 2001). In Australia, weeds have been shown to thrive immediately following fire, increasing both coverage and frequency; a trend that was still evident seven years after the disturbance (Milberg and Lamont 1995). Also in Australia, fires of low intensity were shown to promote spread and abundance of weeds (Briese 1996). In Washington, burning increased exotic species coverage more than threefold, and the number of exotic species continued to increase for three years after the burn (Lehmukuhl 2002). In contrast, Smith and Knapp (1999) found that fire decreased exotic richness and cover.

Thinning could enhance invasions in a number of ways. Seeds of exotic species can be carried on machinery and by people (Harrod 2001), which makes it plausible that exotics could be introduced and spread by the mechanical thinning process itself. Also, thinning reduces competition for light, which could facilitate invasions (Goodwin and

Sheley 2001). Light availability has been shown to be very important in the early stages of exotic establishment, which determined later invasion success (Milbau et al. 2003). The seedbank may account for the continued persistence of many invasive species (Mitchell et al. 1998). Exotic species have been shown to contribute extensively to the seedbank, even when their cover is low in the community, and therefore they may be able to respond strongly to silvicultural thinning (Halpern et al. 1999; Milberg and Lamont 1995). Thinning also reduces vegetative obstructions allowing increased seed dispersal into forests (Cadenasso and Pickett 2001), which could result in higher propagule pressure in thinned treatments. However, Fornwalt et al. (2003) found that longterm land management practices (including logging and prescribed burning) did not have an impact on exotic species abundance.

Ecosystems differ in their susceptibility to invasion (Lonsdale 1999). Those with stress levels exceeding the physiological tolerances of many invaders may be more resistant to invasion (Alpert et al. 2000). However, there is ample evidence this is not the case in ponderosa pine ecosystems. In fact, invasive species have been a land management problem in temperate and boreal coniferous forests for nearly a century (Harrod and Reichard 2001). Open forests, which have frequent breaks in plant cover, feature patches where exotics could establish (Baker 1986). In the Southwest, many exotic species have been documented in ponderosa pine ecosystems after restoration treatments (Sieg et al. 2003). Additionally, Northern Rocky Mountain ponderosa pine forests have been shown to be susceptible to a variety of exotic invaders even without disturbance, although disturbances increase the likelihood of invasion (Weaver et al. 2001; Forcella and Harvey 1983).

Exotics have been defined in a variety of ways in the literature--in contradictory ways (Hettinger 2001). Further complicating the picture is that exotic species are introduced almost continuously, however, the majority of these species do not become problem weeds (Williamson 1996; Radosevich 2003). With many species there is a lag time between introduction, establishment, and emergence as a troublesome weed, making it difficult to predict which species will become problems and which will remain benign (Radosevich et al. 2003). An exotic species may be present in a community for long periods of time without displacing native species until a change in conditions or evolution allows the exotic to become an invader (Alpert et al. 2000; Blossey and Nötzold 1995). Many exotics not considered invasive may still be in the lag phase and could become problems in the future (Hobbs and Humphries 1995). Additionally, non-invasive exotic plants may disrupt the community in such a way that the community becomes more susceptible to invasion by other exotic invasive species (Sher and Hyatt 1999; Grice 2004).

Three sources, Hitchcock and Cronquist (1973), the USDA plant database (<http://plants.usda.gov> 2004), and the invaders database (Rice 2003), were used in this study to determine if plants were exotic or native. Nomenclature follows Hitchcock and Cronquist (1973) except where names have been updated, in which case the USDA plant database was used. Table 1 shows a list of all exotic species present at the study site.

Exotic species differ in terms of the harm they cause. Species that cause significant damage, either economically or environmentally, are given the legal status of noxious (Harrod 2001; Skinner et al. 2000). These species are higher priority for control (Skinner et al. 2000). Knowing species that are problematic in neighboring states

**Table 1:** Exotic species present at the study site, with common and scientific nomenclature and nomenclature authorities from the USDA plants database, and states where the species is declared noxious. Lifespan indicates typical duration of individual plants. See text for description of undesirable species.

<b>Common name</b>	<b>scientific name and authority</b>	<b>lifeform</b>	<b>lifespan</b>	<b>undesirable?</b>	<b>noxious (where)</b>
Red top	<i>Agrostis gigantea</i> Roth	grass	perennial	no	not noxious
Annual bentgrass	<i>Apera interrupta</i> (L.) Beauv.	grass	annual	no	not noxious
Cheatgrass	<i>Bromus tectorum</i> L.	grass	annual	yes	not noxious
Nodding thistle	<i>Carduus nutans</i> L.	forb	biennial/perennial	yes	(ID, OR, WA, WY)
Spotted knapweed	<i>Centaurea biebersteinii</i> DC.	forb	biennial/perennial	yes	(ID, OR, MT, WA, WY)
Common mouse-ear chickweed	<i>Cerastium fontanum</i> Baumg.	forb	biennial/perennial	no	not noxious
Canadian thistle	<i>Cirsium arvense</i> (L.) Scop.	forb	perennial	yes	(ID, OR, MT, WA, WY)
Bull thistle	<i>Cirsium vulgare</i> (Savi) Ten.	forb	biennial	yes	(OR, WA)
Hound's tongue	<i>Cynoglossum officinale</i> L.	forb	biennial	yes	(WA, WY, OR, MT)
Field cottonrose	<i>Logfia arvensis</i> (L.) Holub	forb	annual	no	not noxious
Prickly lettuce	<i>Lactuca serriola</i> L.	forb	annual/biennial	no	not noxious
Yellow sweet clover	<i>Melilotus officinalis</i> (L.) Lam.	forb	annual/biennial/perennial	no	not noxious
Strict forget-me-not	<i>Myosotis stricta</i> Link ex Roemer & J.A. Schultes	forb	annual	no	not noxious
Field timothy	<i>Phleum pratense</i> L.	grass	perennial	no	not noxious
Canadian bluegrass	<i>Poa compressa</i> L.	grass	perennial	no	not noxious
Kentucky bluegrass	<i>Poa pratensis</i> L.	grass	perennial	no	not noxious
Prostrate knotweed	<i>Polygonum aviculare</i> L.	forb	annual/perennial	no	not noxious
Sulphur cinquefoil	<i>Potentilla recta</i> L.	forb	perennial	yes	(MT, OR, WA)
Fragile sour-weed	<i>Rumex acetosella</i> L.		perennial	no	not noxious
Curly dock	<i>Rumex crispus</i> L.	forb	perennial	no	not noxious
Bladder campion	<i>Silene latifolia</i> Poir	forb	biennial/perennial	no	(WA)
Dandelion	<i>Taraxacum officinale</i> G.H. Weber ex Wiggers	forb	perennial	no	not noxious
Field pennycress	<i>Thlaspi arvense</i> L	forb	annual	no	not noxious
Meadow salsify	<i>Tragopogon dubius</i> Scop.	forb	annual/biennial	no	not noxious
Clover	<i>Trifolium</i> spp	forb	perennial	no	not noxious
Common mullein	<i>Verbascum thapsus</i> L.	forb	perennial	yes	Montana county

can assist in determining if a plant should be declared noxious (Skinner et al. 2000). State-level noxious weed lists for the five-state Pacific Northwest region (Wyoming, Montana, Idaho, Washington and Oregon), and Montana county noxious weed lists were used to identify species of concern. If a species occurred on a noxious weed list, then it was considered undesirable. Four species (*Cirsium arvense*, *Carduus nutans*, *Centaurea biebersteinii*, and *Cynoglossum officinale*) in this study are among the 45 species most frequently found on state noxious weed lists (Skinner et al. 2000). Other species considered undesirable are *Bromus tectorum*, *Cirsium vulgare*, *Verbascum thapsus*, and *Potentilla recta*.

Cheatgrass (*Bromus tectorum*) was also included as undesirable because of the extensive damage it has caused in the American West, although it was not on any noxious weed list. This species is an exotic annual that dominates many semiarid areas (Belnap and Phillips 2001). It alters natural fire regimes, and perhaps no other species has so dramatically impacted the region (Harrod and Riechard 2001). Similarly, Pimentel et al. (1999) assert that *Bromus tectorum* is drastically altering the flora and fauna of many natural ecosystems. The impacts extend beyond native plants to soil fungi, invertebrates, and bacteria (Belnap and Phillips 2001).

Restoration treatments are likely to have increased application given the recent passage of the Healthy Forest Restoration Act. However, careful monitoring of treatment effects has rarely occurred after restoration (Block et al. 2001). Given the dearth of information on exotic invasions after restoration, and the detrimental effects exotic plants are inflicting on ecosystems worldwide, this study is particularly timely. This study is unique because treatments were applied at an operational scale in a replicated, balanced,

randomized block design which allows rigorous statistical tests (Waltz et al. 2003; Hulbert et al. 1984; Michner 1997; Block et al. 2001). The FFS studies were designed to evaluate restoration treatments from a broad interdisciplinary perspective to discern which treatment methods are best. This study focuses on one aspect of restoration treatments: exotic invasion.

## Study site

The study site is located at the University of Montana’s Lubrecht Experimental Forest in the Garnet Range of western Montana, at 46.9 degrees north latitude and 113.4 degrees west longitude. Physical characteristics of the site and the habitat type (Pfister et al. 1977) are shown in Table 2. Mean annual temperature is 40.8 degrees Fahrenheit (4.9 degrees Celsius), with 38 cm of annual precipitation (Western Regional Climate Center 2003). The overstory is second-growth ponderosa pine/Douglas-fir (*Pinus ponderosa/Pseudotsuga menziesii*) forest. Fire has been excluded for most of the past century, resulting in vigorous regeneration of

**Table 2:** Treatment unit physical characteristics and habitat type.

<b>Block</b>	<b>Treatment</b>	<b>Soil Series</b>	<b>Habitat Type</b> (Pfister et al. 1977)	<b>Slope</b> <b>(degrees)</b>	<b>Lower</b> <b>Elevation(m)</b>	<b>Upper</b> <b>Elevation (m)</b>	<b>Average</b> <b>Aspect</b>
1	thin-only	Winkler very gravelly sandy loam	<i>Pseudotsuga menziesii/Vaccinium caespitosum</i>	9.5	1275	1337	138.321
1	burn-only	Winkler very gravelly sandy loam	<i>Pseudotsuga menziesii/Vaccinium caespitosum</i>	10.1	1313	1387	126.927
1	thin/burn	Winkler very gravelly sandy loam	<i>Pseudotsuga menziesii/Vaccinium caespitosum</i>	9.8	1337	1388	123.137
1	control	Bignell gravelly loam	<i>Pseudotsuga menziesii/Vaccinium caespitosum</i>	13.5	1288	1351	125.227
2	thin-only	Tevis-Mitten complex	<i>Pseudotsuga menziesii/Vaccinium caespitosum</i>	8.3	1248	1295	2.5411
2	burn-only	Tevis-Mitten complex	<i>Pseudotsuga menziesii/Vaccinium caespitosum</i>	4.9	1275	1304	335.837
2	thin/burn	Tevis-Mitten complex	<i>Pseudotsuga menziesii/Vaccinium caespitosum</i>	8.1	1275	1340	11.0291
2	control	Tevis-Mitten complex	<i>Pseudotsuga menziesii/Vaccinium caespitosum</i>	5.3	1248	1281	74.8616
3	thin-only	Greenough silt loam	<i>Pseudotsuga menziesii/Vaccinium caespitosum</i>	7.8	1240	1284	97.887
3	burn-only	Greenough silt loam	<i>Pseudotsuga menziesii/Spiraea betulifolia</i>	5.1	1259	1277	302.892
3	thin/burn	Greenough silt loam	<i>Pseudotsuga menziesii/Vaccinium caespitosum</i>	6.5	1230	1268	263.242
3	control	Greenough silt loam	<i>Pseudotsuga menziesii/Vaccinium caespitosum</i>	4.6	1244	1263	58.1712

Douglas-fir in the understory (Fiedler et al. 2000). Five tree species are found on site: Douglas-fir, western larch (*Larix occidentalis*), ponderosa pine, lodgepole pine (*Pinus contorta*), and quaking aspen (*Populus tremuloides*). The most abundant undergrowth species include: *Arnica cordifolia*, *Spiraea betulifolia*, *Symphoricarpos albus*, *Calamagrostis rubescens*, and *Carex geyeri*. The primary habitat type according to Pfister et al. (1977) is *Pseudotsuga menziesii/Vaccinium caespitosum* (Douglas-fir/dwarf huckleberry). Soil series found at the site include: Bignell gravelly loam, Winkler very gravelly sandy loam, Tevis-Mitten complex, and Greenough silt loam.

### **Treatments**

The treatment referred to hereafter as “thin-only” was a comprehensive treatment that consisted of an improvement/selection cutting combined with low thinning. The silvicultural prescription was designed to leave a fire resistant stand composed primarily of seral species with good form and vigor. Leave trees were marked to a target reserve basal area of 11.5 m<sup>2</sup> per ha. Large diameter trees of seral species were favored as leave trees, although some seral trees in all size classes were reserved if available. A cut-to-length system was used with a harvester and forwarder. The harvester cut and limbed trees, and sorted product in the field. Slash-piles were left on site and driven on by the machinery to reduce compaction. A rubber-tired, self-loading forwarder moved the product to the landing, so no skidding was involved. Additionally, the work was done in February and early March of 2001, on a snowpack, minimizing soil compaction (data on file, University of Montana, College of Forestry and Conservation, Missoula, MT).

Prescribed broadcast burns were implemented during May and June of 2002, with each of the burn treatment units being burned separately. The relative humidity ranged

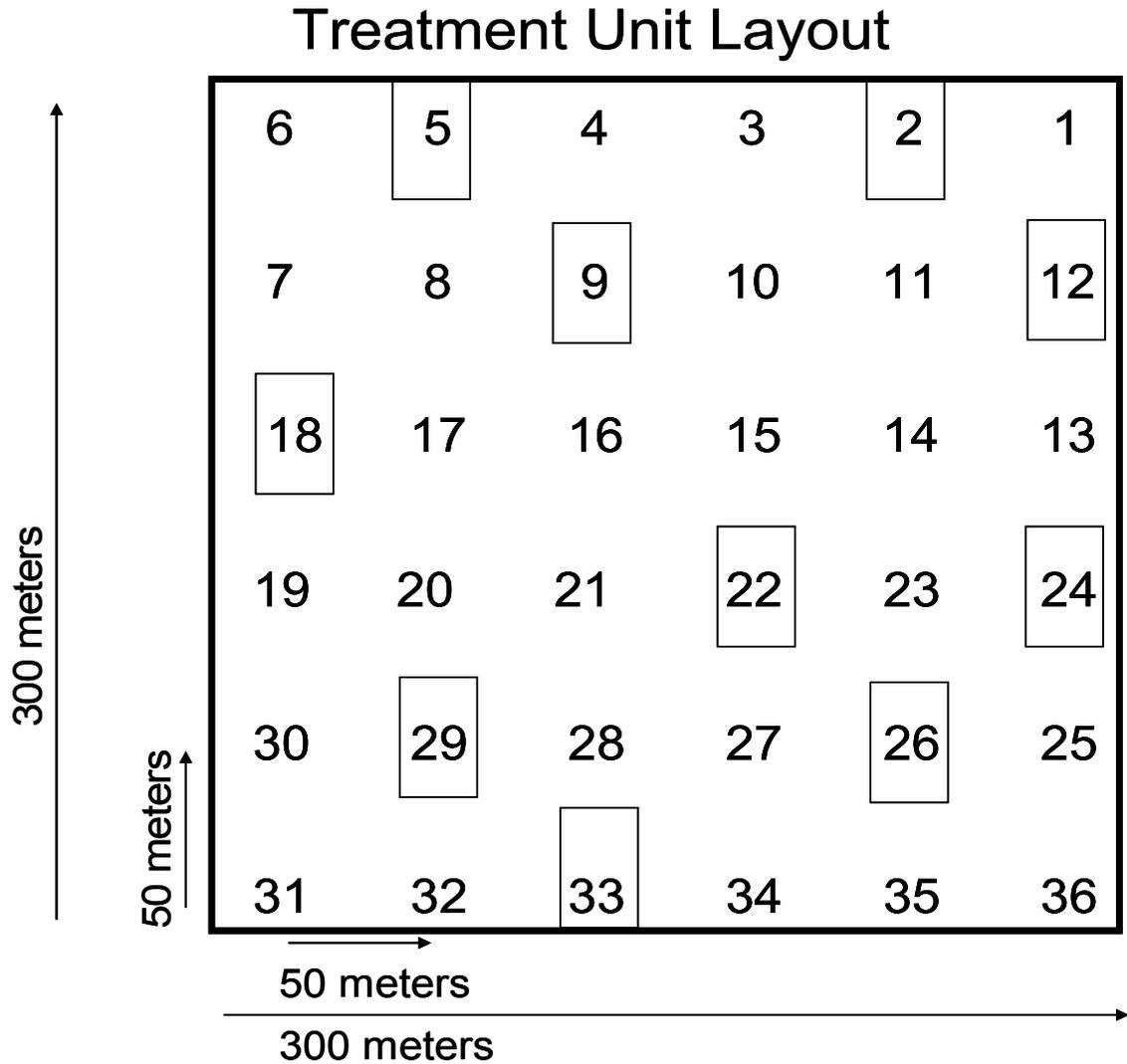
from 20-48 percent, and averaged about 34 percent during the burns. Burn day temperatures ranged from 9-29° C, and averaged around 18° C. Winds were fairly calm, ranging from 2-13 km/hr, although one burn had gusts up to 21 km/hr. An effort was made to keep the burns homogeneous.

### **Field Methods**

Three relatively homogeneous “blocks” of approximately 40 hectares were selected for treatment. Each block was divided into four square treatment units of nine hectares. Each treatment was assigned randomly in each block. Thirty-six points were established in a six x six grid at 50 m intervals within each treatment unit, with all points at least 25 m from the treatment unit edge. Using a stratified random design, ten 50 m x 20 m modified Whittaker plots (0.1 hectare each) were established within each treatment unit, centered at the points. At least one Whittaker plot was established in each row and column but no more than two plots in any row or column, ensuring dispersion throughout the treatment unit (Figure 1). Within each plot, ten 10 m x 10 m square (100 m<sup>2</sup> each) subplots were established. Twelve 1 m x 1 m quadrats (1 m<sup>2</sup>) were located at corners of the subplots using a stratified random design that ensured each subplot had at least one quadrat, and that quadrats were distributed throughout the plot (Figure 2). Plot, subplot, and quadrat corners were permanently marked with rebar to ensure the same location was sampled in future measurements.

After thinning, all overstory trees > 10 cm diameter at breast height (dbh = 1.37 m) present on plots were tagged and identified to species, with dbh measured to the nearest 0.3 cm for each tree. In the thinned treatment units, stumps of cut trees were

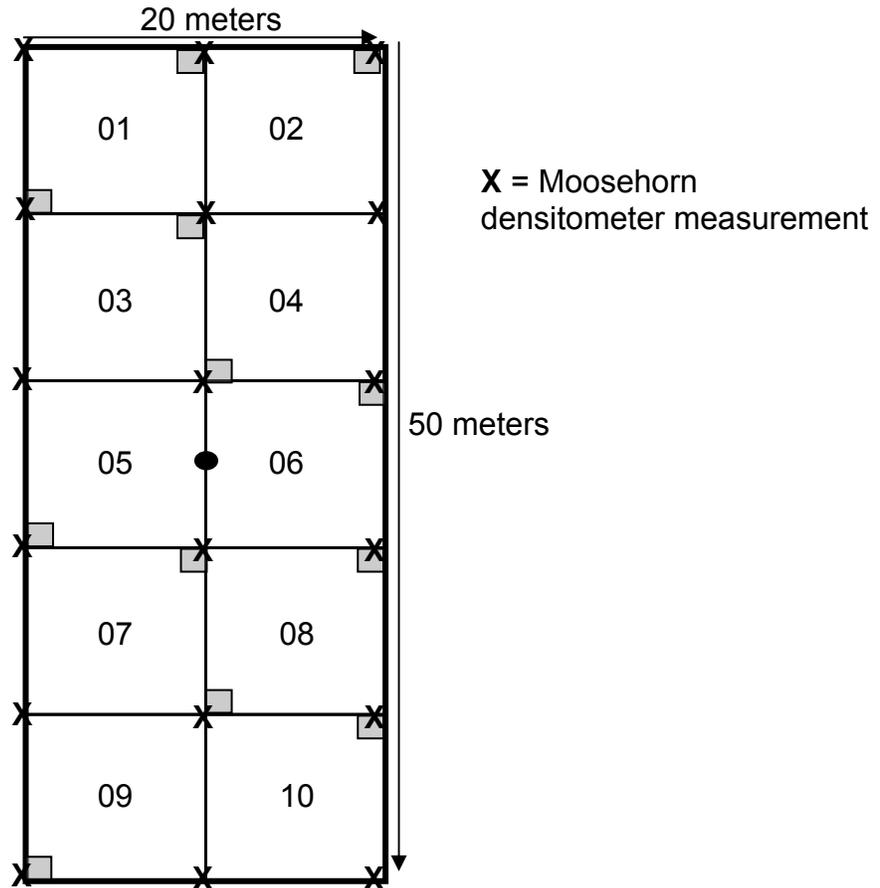
**Figure 1:** Treatment unit layout showing the six by six grid, with modified Whittaker plots highlighted.



measured and stump/dbh regressions were developed from live trees to estimate cut-tree diameters. Mortality was recorded in 2003; however the diameters of living trees were not re-measured. Canopy completeness was estimated using a moosehorn densitometer and canopy cover was estimated by hemispherical fisheye photography. Densitometer measurements were taken at subplot corners (18 per plot--as seen in Figure 2) and live canopy cover was recorded as present if any live foliage was observed and absent otherwise. Fish-eye photographs were taken 1.5 m above the Whittaker plot centers and

percent canopy cover was determined using Gap Light Analyzer (GLA) version 2.0 (Frazer et al. 1999).

**Figure 2:** Modified Whittaker plot layout. Subplots (100m<sup>2</sup>) are numbered and quadrats (1m<sup>2</sup>) are shaded. Hemispherical fish-eye photographs were taken from the plot center and moosehorn densitometer readings were taken at subplot corners.



Saplings were defined as tree regeneration greater than 1.37 meters in height but less than 10 cm in dbh. On each plot, saplings were measured on five subplots selected using a stratified random design. All saplings were identified to species, and three size classes (1 = < 3.0 cm dbh, 2 = >3.0 cm to 6.1 cm dbh, 3 = >6.1 cm to 10.2 cm dbh) were used to classify saplings. Tree and sapling measurements, including trees per hectare, basal area per hectare, and canopy cover/completeness, are shown in Appendix 1.

Understory vegetation cover was ocularly estimated for each plant species at the quadrat level (1 m<sup>2</sup>). A species did not need to be rooted in the quadrat to contribute cover. Percent cover was an estimation of the total area the species would have covered when at maximum aboveground biomass for the year. Pre-treatment cover was estimated using cover codes (0 = 0 percent, 1 = < 1 percent, 2 = 1-10 percent, 3 = 11-25 percent, 4 = 26-50 percent, 5 = 51-75 percent, and 6 = 76-100 percent). Post-treatment cover values were ocularly estimated to the nearest percent. To account for these differences during analysis, the pre-treatment cover code was converted to the median for the cover code, which provided an estimated percent cover. To ensure cover estimate fidelity among sampling years the same person served as crew leader each year. An effort was also made to sample each plot at the same time of year it had been sampled in previous years.

Presence/absence data were collected before and after treatment by recording all species that occurred on the 50 m x 20 m plot. To provide further details in species richness after treatment, all species were recorded on two subplots (03 and 10) selected using a stratified random design. Also, undesirable species cover was sampled on all 10 subplots within a plot, with cover ocularly estimated for each species. A cover value of 0.2 was assigned to species with less than 0.5 percent cover, and estimated to the nearest percent cover thereafter. Pre-treatment vegetation data were collected in the summers of 2000 (thin-only and thin/burn) and 2001 (control and burn-only). Post-treatment vegetation data were collected on all treatments in the summer of 2003. Plants were identified in the field using Hitchcock and Cronquist (1973).

Post-treatment percent cover values were also ocularly estimated at the quadrat level for mineral soil, duff and litter, rock, woody stems (live and dead stems > 1 m in

height), natural wood (downed woody debris large enough to obscure growth), and slash. Duff included non-vascular plants, needles, and highly decayed wood. In 2002, all of the above variables plus char (if burned) were estimated for each quadrat, as well as the intensity of the burn (burn intensity) or mechanical thinning (slash intensity). Treatment intensity was estimated on the following categorical scale: 0 = no change due to treatment, 1 = some change, 2 = significant change on up to half the quadrat, and 3 = profound change on over half of the quadrat. Cover of exposed mineral soil was also estimated at the subplot level in 2003, following the same procedure as described for undesirable weeds. Additionally, slope and aspect were recorded to the nearest degree from the plot center with a compass and clinometer, respectively.

The first fall after the prescribed burns, additional variables related to burn intensity were collected for overstory trees. Percent of crown volume scorched was estimated to the nearest 10 percent except at extreme values, where it was estimated to the nearest five percent. The height of needle scorch was measured with an impulse laser to determine scorch height. If all of the needles on a tree were scorched, a height was not assigned because the scorch height was greater than the tree height. Also, the maximum height of bole char was recorded for each tree.

### **Summary methods**

Shannon's index of diversity (Shannon and Weaver 1949) and species richness (all species--native and exotic) were calculated for each quadrat (120 per treatment unit) before and after treatments were applied. Additionally, the cover for all species combined was calculated for each quadrat both before and after treatment. Changes from

pre-to post-treatment were calculated for exotic species richness, cover, and frequency using Equation 1:

$$[1] \quad \text{change} = (\text{post-treatment minus pre-treatment})$$

Since post-treatment exotic abundance was generally higher than pre-treatment most change values were positive. The change in native species richness and cover were also calculated using Equation 1.

Tree species were combined to estimate basal area (BA), which was calculated for each tree using the following equation:

$$[2] \quad \text{BA(m}^2\text{)} = \text{dbh}^2(\text{cm}) * (0.00007854)$$

The proportional change in BA per plot was then calculated by summing the BA for all trees in a plot, subtracting post-from pre-treatment BA, then dividing by the pre-treatment BA. Trees per ha (tph) was calculated for each plot (0.1 ha) by multiplying the tree count for the plot by 10. The proportional change in tree density was calculated by subtracting post-treatment tree density from pre-treatment density, and dividing the difference by pre-treatment density. Equation 3 was used to calculate canopy completeness from densitometer measurements:

$$[3] \quad \text{Canopy Completeness} = \frac{X}{18} * 100$$

Where X is the number of points where live foliage was observed and 18 is the total number of points for the plot.

All species and size classes of saplings were combined for a single estimate of sapling density per ha. Saplings were tallied on five 0.01 ha subplots on each plot, summed, and then multiplied by 20 to get an estimate of saplings per hectare for each plot. The proportional change in sapling density was calculated for each plot by

subtracting post-treatment sapling density from pre-treatment density, and then dividing by the pre-treatment density.

The aspect data were converted into a linear form following Stage (1976). Using Equation 4, circular values of aspect were converted to a linear dataset

$$[4] \quad \text{Converted aspect} = \cosine ((\text{aspect in degrees}) - 45^\circ)$$

However, the degree of slope also influences the effect of aspect on vegetation (Stage et al. 1976). Therefore, effective aspect was calculated from converted aspect using Equation 5:

$$[5] \quad \text{Effective aspect} = (\text{converted aspect}) * \text{tangent} (\text{angle of slope})$$

Effective aspect was calculated for every plot. Additionally, burn and slash effect variables were calculated by combining the burn (or slash) intensity measure and percent cover of char (or slash) recorded for each quadrat in 2002, using Equation 6:

$$[6] \quad \text{burn effect} = \frac{\text{percent char cover (2002)} * \text{burn intensity (0-3)}}{100}$$

A list of measured and derived environmental variables is displayed in Table 3, and burn related variables are shown in Appendix 2.

### **Analytical Methods**

#### **Non-metric Multidimensional Scaling (NMS)**

Non-metric Multidimensional Scaling (NMS) in PC-Ord 4 (McCune and Mefford 1999) was used to visually examine differences among treatments in exotic species abundance and composition. For this analysis, cover of exotic species at the quadrat-level (1 m<sup>2</sup>) was averaged to the plot level for the 120 Whittaker plots present in the study. Distances between plots could then be calculated based on exotic species cover and composition. The method used for NMS follows that described in Maher (1976) and

Kruskal (1964b). Ordination seeks the strongest structures originally present in high-dimensional space and displays the relationships with as much fidelity as possible in lower dimensions (McCune and Grace 2002). The final result is an optimized map of the plots in ordination (exotic species) space (Kenkel et al. 2002). This provides a simplified visual representation

**Table 3:** List of measured and calculated variables and their descriptions.

<b>Variable</b>	<b>Description</b>
Block	Blocked areas where each treatment was applied
Effective aspect	Combines the effect of aspect and slope (Stage 1976)
Canopy completeness	Percent of points covered by live canopy, from moosehorn densitometer
Canopy cover (GLA)	Percent canopy cover from fisheye photography and GLA (Frazer et al 1999)
Duff	Percent coverage in quadrats of duff and litter
Mineral soil (quadrat)	Percent cover of mineral soil-quadrat
Mineral soil (subplot)	Percent cover of mineral soil-subplot
Rock	Percent coverage of rocks large enough to obscure growth-quadrat
Wood	Percent coverage of naturally occurring wood obscuring growth-quadrat
Woody stem	Percent coverage of woody stems greater than one meter in height-quadrat
Burn intensity	Burn intensity measure (2002)-quadrat
Slash intensity	Slash intensity measure (2002 and 2003)-quadrat
Slash 2002	Percent cover of slash (2002)-quadrat
Slash 2003	Percent cover of slash (2003)-quadrat
Char 2002	Percent cover of char (2002)-quadrat
Burn effect	Percent char cover times burn intensity divided by 100
Slash effect	Percent slash cover times slash intensity divided by 100
Pre cover	Cover of all plant species per quadrat prior to treatment
Cover 2003	Cover of all plant species per quadrat after treatment
Shannon-Weaver pre	Shannon-Weaver (1949) diversity index per quadrat before treatment
Shannon-Weaver 2003	Shannon-Weaver (1949) diversity index per quadrat after treatment
Richness pre	Average pre-treatment quadrat level richness
Richness 2003	Average posttreatment quadrat level richness
Basal area change	Proportional change in overstory basal area (pre-posttreatment/pretreatment)
Native cover change	Change in the cover of native species (pre-posttreatment)
Native richness change	Change in the plot level richness of native species (post-pretreatment)
Change in tree density	Proportional change in overstory trees/ha (pre-posttreatment/pretreatment)
Change in sapling density	Proportional change in saplings/ha (pre-posttreatment/pretreatment)
Percent Crown Scorch	Percent of crown scorched in prescribed burn, per tree
Max bole char	Height in meters of the maximum bole charring, per tree
Max scorch height	Height in meters above ground of the maximum 100% needle scorch, per tree

of the data, enabling researchers to see similarities (and differences) among observations (in this case--plots) based on community composition (Anderson 2001). Plots that are the most similar will appear close together in ordination space (Bond et al. 2002). NMS uses an iterative method based on ranked distances between plots (Waichler et al. 2001);

therefore, unlike many other ordination methods NMS does not depend on linear relationships or multivariate normality of data. Each iteration slightly adjusts the positions of the plots in ordination space seeking the best fit (McCune and Grace 2002). This gives NMS considerable flexibility, and it is generally the best choice for plant community data because such data rarely meet multivariate distributional assumptions (McCune and Grace 2002; Kenkel et al. 2002; Clymo 1980).

Plot-level data were used for this analysis because treatments were not uniform, and the variation within treatments is as interesting as the central tendency (Underwood 1997). NMS cannot be performed with plots that have zero cover for all exotic species because the algorithm requires a positive distance between all plots. Therefore, plots that did not have any exotics were removed from the dataset for this analysis. Thus 22 plots were deleted and 98 plots remained. The number of plots deleted differed by treatment (one in the thin/burn, five in the thin-only, seven in the burn-only, and nine in the control) so the dataset was unbalanced. A Euclidean distance measure was used to keep the distance measure in NMS and Multi-Response Permutation Procedures (MRPP) equivalent (McCune and Grace 2002).

Three measures were used to estimate how well the NMS ordination represented the data in the original high-dimensional space. The coefficient of determination ( $R^2$ ) between the distances in the original space and the ordination space provides information on how much variation is represented by each of the axes, and by the ordination as a whole (cumulative). Stress is a measure of the departure from monotonicity in the relationship between plot distances in the original 21-species space and their distance in the reduced ordination space (Waichler et al. 2001), so lower values indicate a better

ordination. Stress decreases with increasing axes, with a stress of zero being possible with one less axes than the number of variables (Kruskal 1964a). Finally, it is important to minimize instability, which is defined as the standard deviation in stress over the past 10 iterations (McCune and Grace 2002). This prevents getting a low value of stress for an ordination that is not stable (see McCune and Grace 2002 for further discussion on instability).

In NMS the number of axes (dimensions) produced is specified by the user. The number of axes that would produce the best ordination was determined by running an automated NMS in PC-Ord 4 with a step down from six axes to one (McCune and Mefford 1999). The final run had two dimensions, with a maximum of 250 iterations and a stability criterion of 0.0005. Fifty runs were conducted with real data. A random starting configuration was used with time of day as the random number generator, and 250 iterations to determine stability. Fifty Monte Carlo runs were used to determine the probability of an equal or lower stress due only to chance. This was a test of whether the structure in the original data matrix was stronger than that expected by chance (McCune and Grace 2002).

#### Blocked Multi-Response Permutation Procedures (BMRPP)

For formal statistical comparisons, data were averaged to the treatment unit level ( $n = 12$  observations--three for each treatment). This provided statistically independent data as each prescribed burn and thinning treatment were considered treatment replicates, which avoided pseudoreplication (Hurlbert 1984). Where possible the change in exotic richness, cover, or frequency was used (all except the subplot level). Blocked Multi-Response Permutation Procedures (BMRPP) were used to test the null hypothesis of no

differences among treatments. MRPP is part of a larger family of techniques that do tests based on the measure of distance (or dissimilarity) between pairs of observations (see Anderson 2001). Even though transforming the data by using the change from pre-to post-treatment should help normalize the data (McCune and Grace 2002), there was still moderate to strong evidence of heteroscedacity. Therefore, nonparametric procedures were used. MRPP is a powerful tool for community data because assumptions of normality and homoscedacity-- required by parametric procedures, some non-parametric tests and rarely met with community data--are not required (McCune and Grace 2002; Anderson 2001). The purpose of MRPP is to detect concentrations within treatments, similar to a parametric student-t or F test. In MRPP this is accomplished by comparing pair-wise distances within a group with the average pair-wise distance for all observations (Zimmerman et al. 1985).

To perform the test--delta--the linear combination of average pair-wise within-group distance was calculated (Zimmerman et al. 1985). Results were then compared with all other possible deltas by using a permutation procedure to randomize the data within groups or treatments (Mallik et al. 2001). The p-value is the probability, under the null hypothesis of no differences among treatments, of obtaining a delta as small or smaller than the observed delta with the random permutations (Mielke 1991). Since most data sets (including the one in this study) have enough observations to make calculating the exact p-value difficult, the following T-statistic is calculated [Equation 7]:

$$[7] \quad T = \frac{\text{observed delta} - \text{expected delta}}{\text{standard deviation of the expected delta}}$$

The observed delta is the average within-group (here within-treatment) distance, while the expected delta is the mean distance for all possible partitions of the data. Therefore,

T describes the separation between groups (McCune and Grace 2002). This test statistic follows a Pearson type III distribution that allows a convenient estimation of a p-value (Zimmerman et al. 1985). An additional estimate of effect size is given by the chance corrected within group agreement (A), which is calculated according to Equation 8:

$$[8] \quad A = 1 - (\text{observed treatment mean} / \text{expected mean (average pair-wise distance)})$$

If all observations within a treatment are identical, A would be equal to one, but if the observed mean equals the expected (treatment does not make observations closer than expected by chance), then A is zero (McCune and Grace 2002). For further details see McCune and Grace (2002) and Zimmerman et al. (1985).

MRPP is not able to partition variation across treatments, making it impossible to discern which treatments are significantly different (Anderson 2001). Pair-wise comparisons between treatments were not done due to the small sample size (n=3) and because of the potential dangers associated with pair-wise comparisons (McCune and Grace 2002). Examining which treatments were different when MRPP detected significant differences was left to be determined by examining which treatments had the most (and least) mean cover, frequency, and richness. Also, differences among treatments were assessed visually by Non-metric Multidimensional Scaling (NMS), and statistically with Indicator Species Analysis, which was used to detect treatments that favored specific exotic species.

BMRPP analysis was performed in PC-Ord 4 (McCune and Mefford 1999) with a Euclidean distance measure. Sorenson distance, which has been highly advocated for ecological data sets (McCune and Grace 2002), could not be used because change data from pre-to post treatment resulted in some negative values. However, Euclidean

distance is preferable for MRPP to Euclidean squared distance (Mielke 1991; Zimmerman et al. 1985), so it was chosen.

Blocked MRPP (BMRPP) tests were used because of the randomized block experimental design. The block median was subtracted from each treatment unit prior to testing for treatment differences (McCune and Grace 2002). This median alignment focuses the analysis on differences among treatments (Zimmerman et al. 1985). With the blocked analysis the calculation of delta is performed according to Equation 9:

$$[9] \quad \text{blocked delta} = \left[ g \binom{b}{2} \right]^{-1} \sum_{i=1}^g \sum_{j < k} \Delta(x_{ij}, x_{ik}),$$

where there are b blocks and g groups (treatments).

For a more detailed discussion of how blocking alters the statistical testing see McCune and Grace (2002), and Mielke and Iyer (1982).

For BMRPP tests, cover, frequency, and richness data for both undesirable and exotic species were averaged to the treatment unit level ( $n = 12$ ) at all three scales. Species were tested individually, and combined into undesirable and exotic groups by summing the cover of all the species in each group respectively. Richness data were tested at all three scales ( $1 \text{ m}^2$ ,  $100 \text{ m}^2$ , and  $1000 \text{ m}^2$ ) for undesirable and exotic species, while frequency was tested at all three scales for undesirable species, and at the quadrat and plot level for other exotics. Cover data from the quadrat level were tested for all exotic species, while undesirable species were also tested at the subplot level. Subplot-level data were tested for differences among treatments, while other scales were tested for differences among treatments in the change from pre-to post-treatment.

Restoration treatments performed at larger scales often have small sample sizes. In several recent restoration studies p-values of  $< 0.1$  have been accepted as significant

(Skov et al. 2004; Kaye and Hart 1998; Waltz et al. 2003). For the BMRPP tests, which had 12 observations (three for each treatment), a p-value of  $< 0.1$  was accepted as marginally significant while a p-value of  $< 0.05$  was considered significant, which allows the reader some interpretation.

### Indicator Species Analysis

To test which treatments may favor specific exotic species, an Indicator Species Analysis was performed. Coverage data of all exotic species from the quadrats ( $1 \text{ m}^2$ ) were averaged to the plot level. The average percent cover for a plot was used instead of averaging to the treatment unit level to provide a more accurate estimate of how frequently a species was found in a given treatment. Analysis was performed in PC-Ord4 (McCune and Mefford 1999) following the technique described in Dufrene and Legendre (1997) and McCune and Grace (2002). This technique produces an indicator value (IV) for every species based on the percent cover of the species in a treatment, and the number of plots the species occurs on within the treatment where it has the greatest cover (Dufrene and Legendre 1997). The fraction of cover in the treatment with the greatest cover for a species is multiplied by the frequency (here number of plots out of 30) where the species is found to calculate the IV. This technique assigns an IV to every species individually, so it is useful for comparing how a particular species was affected by different treatments (McCune and Grace 2002).

The IV ranges from 0 to 100--with a 100 being a perfect indicator, which means the species is found only in one treatment and in all plots of that treatment. The p-value was calculated by a Monte Carlo test with 1000 runs and time of day as the random number seed source. The p-value is the probability of randomly obtaining an IV value

(by shuffling observations among plots) that is equal to or exceeds that calculated by using treatment groups to divide the sample (McCune and Grace 2002; Dufrene and Legendre 1997). Each treatment was compared in a pair-wise fashion with the control because two treatments (e.g., thin/burn and burn-only) could have had a similar impact on the cover of an exotic species where both were different than the control, but not different from one another, making the IV insignificant. Because many of the exotics were relatively rare, a cut-off level of 20 was used for accepting an IV as being meaningful (slightly lower than the value of 25 used by Dufrene and Legendre 1997), if in addition the p-value was  $< 0.05$ .

#### Covariate Analysis

Thinning and prescribed burning facilitate invasions in some cases but not in others, so environmental variables were analyzed using univariate regression in a general linear model (SPSS version 11.0) to determine which may be associated with invasion. Plots varied in physical characteristics and treatment intensities. Yet treatment unit averages may not detect the extremes of environment and treatment most important in determining invasion (Underwood 1997). Therefore, although not (rigorously) statistically independent samples, plot level ( $n = 120$ ) data were used to assess the association of environmental factors with exotic invasions.

It is important to understand what aspects of the environment contribute to invasion success to curtail invasions in the future. This requires focusing on processes and structures that facilitate invasions, and not on individual species (Hobbs and Humphries 1995; McEvoy and Coombs 1999). Exotic richness provides an estimate that weights all species equally, and is resistant to observer error. Richness is dependent on

the area measured, but the balanced design of this study ensures the same area is measured for each treatment. Pre-treatment data provide initial differences among plots and treatments, so that the effect of treatment and environmental variables is clearer. Therefore, the change in plot level (1000 m<sup>2</sup>) exotic species richness (n = 120) from pre- to post-treatment was used as the response variable to determine what environmental attributes are associated with invasion.

All environmental variables were summarized to the plot level (n = 120) for regression analyses even if not measured at that scale originally. Levene's test for homogeneity of variance, and normal probability-probability plots (p-p plots) were used to assess assumptions of normality and equal variance. First a regression with only treatment, block, and their interaction was performed, treating both as factors. To derive the most parsimonious model, terms were eliminated from the model in a stepwise fashion if the extra sums of squares F-test was not significant, starting with higher order terms (interaction), then removing main effects if they were not significant themselves or in an interaction.

For the second regression analysis, treatment was removed from the model because of high correlation with several of the environmental variables. This allowed detection of which environmental variables associated with treatment are important in determining invasion success. Variables available for analysis are shown in Table 3. For variables measured in multiple ways (i.e., mineral soil and canopy cover/completeness) only the variable with the most evidence (lowest p-value) for the coefficient ( $\beta$ ) not being equal to zero was included in the model. Additionally, because burn and slash effect were calculated from other variables, only the most significant from each group was

included in the model. First each variable was input individually into the model. Block was treated as a factor and the extra sums of squares F-test was used to determine if it was significant. Then all significant variables were input into the model as a group. To derive a parsimonious model, all variables whose  $\beta$  was not significantly ( $p > 0.05$ ) different than zero (or the extra sums of squares F-test for block) were eliminated from the model in a step-wise fashion until only significant variables remained. Parameter estimates for the final model were then generated. Due to the large number of variables, interaction terms were not examined.

Once a final regression model was derived, significant ( $p < 0.05$ ) explanatory variables were averaged to the treatment unit level ( $n = 12$ ) to test for treatment differences. Environmental variables are often more normal in distribution than community data (Kenkel et al. 2002), so univariate regressions were performed in SPSS (version 11.0) with treatment and block as factors. The extra sums of squares F-test was used to test for significance of block and treatment in predicting values for each environmental response variable. Levene's test for heterogeneity and plots of predicted versus residual values were used to test the assumptions of equal variance. Normal p-p plots were used to assess the normal distribution assumption. Where block was not significant, it was eliminated from the model. Where treatment was significant, post hoc least significant difference (LSD) tests were used to determine which treatments were significantly different. Due to the pair-wise comparisons between treatments in the LSD tests, a p-value of  $< 0.05$  was accepted as significant despite the small sample size.

## Results

### Exotic Plant Community Patterns

Non-metric multidimensional scaling (NMS) provided a visualization of the exotic plant community in each plot, with plots that were close together being similar in their exotic communities. The stress for the NMS ordination was 5.27, which constitutes a good ordination with little chance for misinterpreting the data (McCune and Grace 2002; Clarke 1993). The Monte Carlo test with 50 runs of randomized data revealed that the probability of obtaining such a small stress randomly was  $p = 0.02$ . The final instability of the ordination was 0.0003. Figure 3 shows the graph of stress versus the number of axes for the step down in dimensionality ordination done to determine the optimum number of axes. Beyond two axes the stress decreased only slightly, therefore the final ordination had two axes. Also, the stress was less for the real data than expected by chance no matter what the number of axes, revealing that there is an underlying structure in the data matrix. The coefficient of determination ( $R^2$ ) for the correlation of distances in the original matrix and the final ordination space was 0.607 for the first axes and 0.331 for the second, for a total of 0.938.

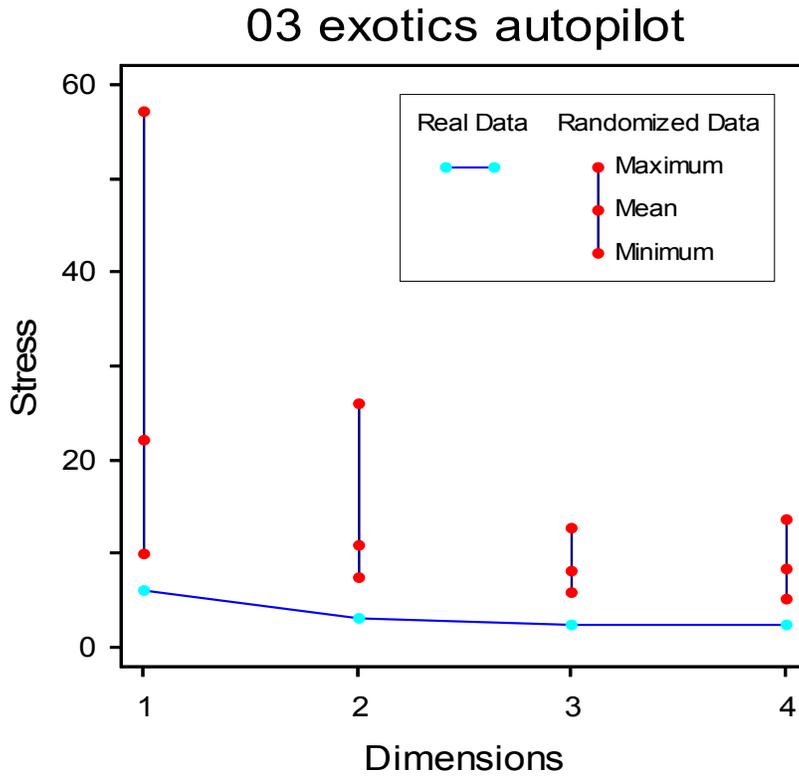
Figure 4 shows the arrangement of plots in exotic species space. The control plots (21 total) were all very similar in ordination space. The burn-only plots were also very similar to each other and the control, while a few thin-only plots showed some differences in exotic species composition and abundance. The thin/burn had over a half-dozen plots that differed from any of the other treatments in the ordination. Additionally, the thin/burn had the greatest variability of any of the treatments, indicating that it is different and much more variable than the other treatments in regard to the exotic species

community. Figure 5 plots scores in ordination space (same scale as Figure 4), for the seven species determined to be at least marginally different ( $p < 0.1$ ) in cover among treatments by BMRPP. The three thistle species (*Cirsium arvense*, *Cirsium vulgare*, and *Carduus nutans*), *Trifolium* species, and *Verbascum thapsus* seem to be driving differences observed in the thin/burn, while *Poa pratensis* and *Logfia arvensis* are not as important.

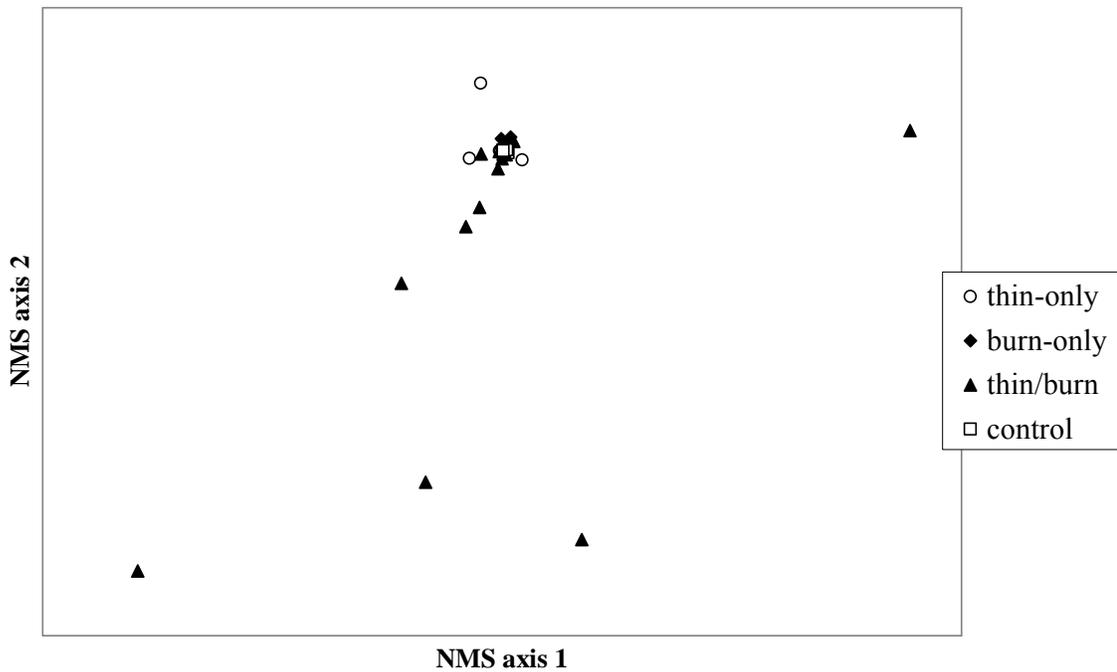
Blocked Multi-Response Permutation Procedures (BMRPP) were used to evaluate the average treatment unit ( $n = 12$ ) exotic abundance variables at multiple scales. Results for change in quadrat-level ( $1 \text{ m}^2$ ) cover for each exotic species and for exotic and undesirable species combined are shown in Table 4 with the p-value for the test of no treatment differences, and A (the chance corrected within group agreement) giving an estimate of treatment effect. The mean change in cover for each treatment, and the standard error of the mean (from the three replicates per treatment) are also displayed. Mean cover change provides insight into which treatments may be causing the observed differences, because BMRPP does not partition variance that is needed to conduct post hoc pair-wise comparisons.

Combined changes in the treatment unit average quadrat-level cover for all undesirable and exotic species were significantly different among treatments ( $p = 0.018$  for exotics and  $p = 0.013$  for undesirable species). The thin-only had a slight decrease in average cover of all exotics (-0.118 percent, standard error  $\pm 0.189$ ); however, the value plus one standard error included zero. The undesirable cover in the thin-only also decreased by an average of 0.206 ( $\pm 0.194$ ) percent per quadrat after treatment. For the

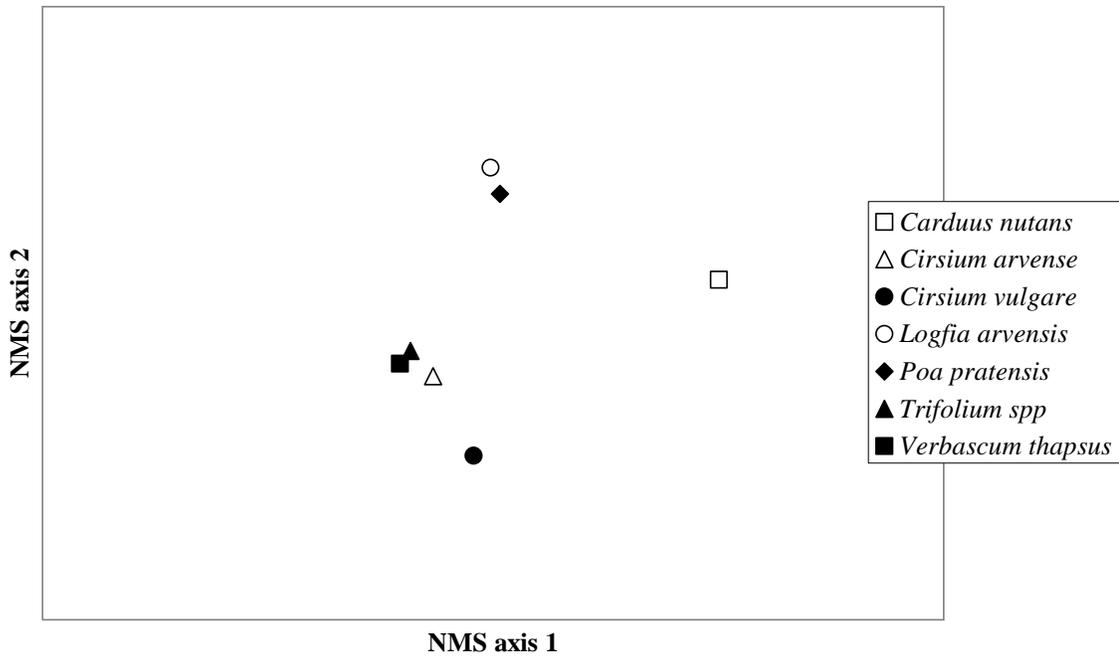
**Figure 3:** Plot of stress versus number of axes for initial NMS run with real and randomized data.



**Figure 4:** Scatterplot of the scores for the 98 plots in which exotic species were found after treatment, from the NMS ordination.



**Figure 5:** Scatterplot of the NMS ordination scores for each species determined to be significantly different among treatments by BMRPP.



undesirable cover, adding one standard error from the mean does not quite include zero, which indicates that the overall decrease in exotic cover in the thin-only is due to decreased cover of undesirable species. The primary species causing these decreases was *Centaurea biebersteinii*--with an average percent cover decrease of 0.199 (+/- 0.203). The three other treatments all showed an average increase in cover for both exotic and undesirable species from pre-to post-treatment. The thin/burn had the largest increases-- 1.747 (+/- 0.951) percent for all exotics and 1.145 (+/- 0.680) percent for undesirable species. The control treatment increased by an average of 0.100 (+/- 0.034) percent in exotic cover, and 0.014 (+/- 0.012) percent for undesirable cover. The burn-only had an average increase of 0.197 (+/- 0.142) percent in exotic cover and 0.085 (+/- 0.077) percent in undesirable cover. Overall 21 exotic species were found at the quadrat level.

Of these species, 16 had their greatest increase from pre-to post-treatment in cover in the thin/burn treatment, while none increased the most in the control.

Seven species had changes in quadrat-level coverage that were at least marginally significantly different among treatments, with four of these species being undesirable. The most significant change in cover was for the undesirable weed *Verbascum thapsus*, with a chance corrected within group agreement of 0.523 and a p-value of 0.014. For this species, the average quadrat level cover increased by 0.646 (+/-0.242) percent in the thin/burn treatment. The control showed no increase while the other treatments had only slight increases (no more than 0.013 percent). The other three undesirable species that had at least marginally significant increases in cover were all thistles (*Cirsium arvense*, *Cirsium vulgare*, and *Carduus nutans*). All three of these species had the greatest increase in percent cover within the thin/burn--0.083 (+/- 0.079), 0.151 (+/- 0.139), and 0.200 (+/- 0.192) percent respectively-- while none had an increase of more than 0.001 percent in any other treatment. *Trifolium* species and *Poa pratensis* were two non-undesirable exotics that also showed the greatest increase in percent cover in the thin/burn. For both of these species, the BMRPP test for differences among treatments was marginally significant ( $p < 0.1$ ). *Logfia arvensis* was the only other species that showed a significant change ( $p = 0.033$ ) in cover from pre-to post treatment. It increased equally in both the thin/burn and burn-only treatments (adding an average of 0.025 percent cover in each), while it did not increase in either the thin-only or control.

The average treatment unit ( $n = 12$ ) change in quadrat-level frequency of all exotic species, and the change in overall richness of undesirable and exotic species, are shown in Table 5. BMRPP results for the test of the null hypothesis of no differences

among treatments are included along with mean frequency and richness changes and standard errors. Changes in quadrat-level frequency revealed similar trends to cover. Both overall exotic and undesirable species richness (average number of exotic or undesirable species per quadrat) were significantly different among treatments ( $p = 0.015$  and  $p = 0.011$  respectively). The thin/burn treatment had the largest change in richness of exotic species with an average of 0.458 ( $\pm 0.126$ ) new exotic species occurrences per  $m^2$  sampled, compared to only 0.033 ( $\pm 0.032$ ) for the control. The same pattern was evident with undesirable species, as the thin/burn had the greatest increase in undesirable richness, 0.183 ( $\pm 0.055$ ), and the control had the least, 0.003 ( $\pm 0.011$ ). Overall, 18 of the 21 exotic species showed the largest or tied for the largest increase in frequency from pre-to post-treatment in the thin/burn, while only *Phleum pratense* increased the most in the control.

For six individual species, there were at least marginally significant differences among treatment. *Carduus nutans* and *Cirsium vulgare* increased with 0.022 ( $\pm 0.015$ ) and 0.019 ( $\pm 0.007$ ) new occurrences per  $m^2$ , respectively, only within the thin/burn treatment. The control and burn-only showed no change in either thistle species, while the thin-only had no change in *C. vulgare* and a slight decline in *C. nutans*. *Logfia arvensis* was not found prior to treatment and only in the burned treatments afterward (at this level). Differences among treatments were significant ( $p = 0.033$ ). The greatest increase was in the thin/burn treatment with 0.025 ( $\pm 0.005$ ) new occurrences per  $m^2$ .

**Table 4:** Mean and standard errors (SE) for treatment (n = 12) change in quadrat-level cover of exotic species. Undesirable species in **bold**. Blocked Multi-Response Permutation Procedures (BMRPP) were used to test the null hypothesis of no differences among treatments. A is the chance corrected within group agreement. \* indicates marginally significant (p < 0.1) and \*\* significant (p < 0.05)

Treatment	thin-only		burn-only		thin/burn		control		BMRPP results	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	A	p
<i>Agrostis gigantea</i>	0.011	0.011	-0.008	0.025	0.008	0.005	-0.003	0.003	-0.083	0.718
<i>Apera interupta</i>	0.003	0.003	0.000	0.000	0.010	0.008	0.000	0.000	0.000	0.446
<b><i>Bromus tectorum</i></b>	<b>0.000</b>	<b>0.000</b>	<b>-0.001</b>	<b>0.001</b>	<b>0.010</b>	<b>0.010</b>	<b>-0.003</b>	<b>0.001</b>	<b>0.053</b>	<b>0.110</b>
<i>Carduus nutans</i>	<b>-0.001</b>	<b>0.001</b>	<b>0.000</b>	<b>0.000</b>	<b>0.200</b>	<b>0.192</b>	<b>0.000</b>	<b>0.000</b>	<b>0.021</b>	<b>0.062*</b>
<i>Centaurea biebersteinii</i>	<b>-0.199</b>	<b>0.203</b>	<b>0.028</b>	<b>0.024</b>	<b>0.019</b>	<b>0.007</b>	<b>0.011</b>	<b>0.011</b>	<b>-0.011</b>	<b>0.671</b>
<i>Cirsium arvense</i>	<b>-0.019</b>	<b>0.019</b>	<b>0.000</b>	<b>0.000</b>	<b>0.083</b>	<b>0.079</b>	<b>0.001</b>	<b>0.001</b>	<b>0.022</b>	<b>0.055*</b>
<i>Cirsium vulgare</i>	<b>0.001</b>	<b>0.001</b>	<b>0.000</b>	<b>0.000</b>	<b>0.151</b>	<b>0.139</b>	<b>0.000</b>	<b>0.000</b>	<b>0.046</b>	<b>.021**</b>
<i>Cynoglossum officinale</i>	<b>0.003</b>	<b>0.003</b>	<b>0.000</b>	<b>0.000</b>	<b>0.000</b>	<b>0.000</b>	<b>0.000</b>	<b>0.000</b>	<b>0.000</b>	<b>0.500</b>
<i>Logfia arvensis</i>	0.000	0.000	0.025	0.017	0.025	0.005	0.000	0.000	0.321	0.033**
<i>Lactuca serriola</i>	0.000	0.000	0.006	0.006	0.000	0.000	0.000	0.000	0.000	0.500
<i>Myosotis stricta</i>	0.003	0.003	0.003	0.003	0.022	0.012	0.000	0.000	0.176	0.110
<i>Phleum pratense</i>	-0.021	0.021	0.003	0.003	0.021	0.012	0.017	0.015	0.055	0.263
<i>Poa compressa</i>	0.004	0.004	0.003	0.003	0.011	0.007	0.003	0.003	-0.026	0.574
<i>Poa pratensis</i>	-0.001	0.035	0.010	0.005	0.082	0.027	-0.007	0.009	0.208	0.093*
<i>Polygonum aviculare</i>	0.000	0.000	0.000	0.000	0.003	0.003	0.000	0.000	0.000	0.500
<b><i>Potentilla recta</i></b>	<b>-0.003</b>	<b>0.010</b>	<b>0.056</b>	<b>0.053</b>	<b>0.035</b>	<b>0.027</b>	<b>0.004</b>	<b>0.016</b>	<b>0.044</b>	<b>0.242</b>
<i>Rumex acetosella</i>	0.000	0.000	0.000	0.000	0.006	0.006	0.000	0.000	0.000	0.500
<i>Taraxacum officinale</i>	0.047	0.015	0.053	0.040	0.106	0.032	0.063	0.031	0.097	0.248
<i>Tragopogon dubius</i>	0.006	0.003	0.007	0.004	0.014	0.010	0.010	0.010	-0.176	0.854
<i>Trifolium</i> species	0.036	0.038	0.013	0.013	0.296	0.201	0.004	0.011	0.092	0.097*
<b><i>Verbascum thapsus</i></b>	<b>0.013</b>	<b>0.010</b>	<b>0.003</b>	<b>0.003</b>	<b>0.646</b>	<b>0.242</b>	<b>0.000</b>	<b>0.000</b>	<b>0.523</b>	<b>0.014**</b>
Overall Exotic Cover	-0.118	0.189	0.197	0.142	1.747	0.951	0.100	0.034	0.253	0.018**
<b>Overall Undesirable Cover</b>	<b>-0.206</b>	<b>0.194</b>	<b>0.085</b>	<b>0.077</b>	<b>1.145</b>	<b>0.680</b>	<b>0.014</b>	<b>0.012</b>	<b>0.252</b>	<b>0.013**</b>

**Table 5:** Mean and standard errors (SE) for treatment (n = 12) change in quadrat-level frequency of exotic species. Undesirable species in **bold**. Blocked Multi-Response Permutation Procedures (BMRPP) were used to test the null hypothesis of no differences among treatments. A is the chance corrected within group agreement. \* indicates marginally significant (p < 0.1), and \*\* significant (p < .05).

Treatment	thin-only		burn-only		thin/burn		control		BMRPP results	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	A	p
<i>Agrostis gigantea</i>	0.006	0.006	0.000	0.017	0.008	0.005	-0.006	0.006	0.030	0.392
<i>Apera interrupta</i>	0.003	0.003	0.000	0.000	0.008	0.008	0.000	0.000	-0.067	0.677
<b><i>Bromus tectorum</i></b>	<b>0.000</b>	<b>0.000</b>	<b>-0.003</b>	<b>0.003</b>	<b>0.008</b>	<b>0.008</b>	<b>-0.006</b>	<b>0.003</b>	<b>0.091</b>	<b>0.110</b>
<i>Carduus nutans</i>	<b>-0.003</b>	<b>0.003</b>	<b>0.000</b>	<b>0.000</b>	<b>0.022</b>	<b>0.015</b>	<b>0.000</b>	<b>0.000</b>	<b>0.176</b>	<b>0.062*</b>
<i>Centaurea biebersteinii</i>	<b>0.008</b>	<b>0.005</b>	<b>0.008</b>	<b>0.005</b>	<b>0.006</b>	<b>0.003</b>	<b>0.000</b>	<b>0.000</b>	<b>0.000</b>	<b>0.456</b>
<i>Cirsium arvense</i>	<b>0.003</b>	<b>0.003</b>	<b>0.000</b>	<b>0.000</b>	<b>0.008</b>	<b>0.005</b>	<b>0.000</b>	<b>0.000</b>	<b>0.067</b>	<b>0.158</b>
<i>Cirsium vulgare</i>	<b>0.000</b>	<b>0.000</b>	<b>0.000</b>	<b>0.000</b>	<b>0.019</b>	<b>0.007</b>	<b>0.000</b>	<b>0.000</b>	<b>0.500</b>	<b>0.016**</b>
<i>Cynoglossum officinale</i>	<b>0.003</b>	<b>0.003</b>	<b>0.000</b>	<b>0.000</b>	<b>0.000</b>	<b>0.000</b>	<b>0.000</b>	<b>0.000</b>	NA <sup>a</sup>	NA <sup>a</sup>
<i>Logfia arvensis</i>	0.000	0.000	0.025	0.017	0.025	0.005	0.000	0.000	0.321	0.033**
<i>Lactuca serriola</i>	0.000	0.000	0.006	0.006	0.000	0.000	0.000	0.000	NA <sup>a</sup>	NA <sup>a</sup>
<i>Myosotis stricta</i>	0.003	0.003	0.003	0.003	0.022	0.012	0.000	0.000	0.176	0.110
<i>Phleum pratense</i>	0.003	0.003	0.003	0.003	0.011	0.007	0.011	0.011	-0.143	0.815
<i>Poa compressa</i>	0.003	0.003	0.003	0.003	0.011	0.007	0.003	0.003	0.000	0.500
<i>Poa pratensis</i>	0.033	0.000	-0.006	0.006	0.064	0.026	0.000	0.000	0.434	0.010**
<i>Polygonum aviculare</i>	0.000	0.000	0.000	0.000	0.003	0.003	0.000	0.000	NA <sup>a</sup>	NA <sup>a</sup>
<b><i>Potentilla recta</i></b>	<b>0.008</b>	<b>0.008</b>	<b>0.017</b>	<b>0.017</b>	<b>0.019</b>	<b>0.012</b>	<b>0.008</b>	<b>0.008</b>	<b>0.040</b>	<b>0.353</b>
<i>Rumex acetosella</i>	0.000	0.000	0.000	0.000	0.006	0.006	0.000	0.000	NA <sup>a</sup>	NA <sup>a</sup>
<i>Taraxacum officinale</i>	0.014	0.018	0.008	0.019	0.069	0.017	0.014	0.010	0.218	0.060*
<i>Tragopogon dubius</i>	0.003	0.003	0.006	0.003	0.014	0.010	0.000	0.000	-0.020	0.532
<i>Trifolium</i> species	0.011	0.015	0.011	0.011	0.033	0.022	0.008	0.005	-0.062	0.691
<b><i>Verbascum thapsus</i></b>	<b>0.006</b>	<b>0.006</b>	<b>0.003</b>	<b>0.003</b>	<b>0.100</b>	<b>0.025</b>	<b>0.000</b>	<b>0.000</b>	<b>0.570</b>	<b>0.015**</b>
Overall Exotic Cover	0.103	0.042	0.083	0.051	0.458	0.126	0.033	0.032	0.504	0.015**
<b>Overall Undesirable Cover</b>	<b>0.025</b>	<b>0.017</b>	<b>0.025</b>	<b>0.017</b>	<b>0.183</b>	<b>0.055</b>	<b>0.003</b>	<b>0.011</b>	<b>0.623</b>	<b>0.011**</b>

a The skewness of delta approaches infinity, so calculations cannot be done. These species were found in no more than 2 plots out of a possible 120.

*Poa pratensis*, while only marginally different in change in cover was significantly different in frequency ( $p = 0.010$ ). Frequency in the thin/burn increased by 0.064 ( $\pm 0.026$ ) occurrences per  $m^2$ , while the control showed no increase. *Taraxacum officinale*, which was not different among treatments in cover, was marginally different in change in frequency ( $p = 0.060$ ). The greatest increase in frequency was in the thin/burn, with 0.069 ( $\pm 0.017$ ) more occurrences per  $m^2$ , while the burn-only had the smallest increase with 0.014 ( $\pm 0.010$ ) new occurrences per  $m^2$ . *Verbascum thapsus* was the only other species that exhibited significant change in frequency among treatments ( $p = 0.015$ ), with no change in occurrences in the control and an increase of 0.1 ( $\pm 0.025$ ) new occurrences per  $m^2$  in the thin/burn.

The average treatment unit level change in exotic species frequency at the plot-level ( $1000 m^2$ ) and overall changes in richness for exotic and undesirable species at the same level, are shown in Table 6. Changes in undesirable and exotic richness were significantly different among treatments ( $p$ -values of 0.002 and 0.003, respectively). All treatments (including the control) increased in exotic richness after treatments were applied. However, the thin/burn added the highest average additional number of exotic species per plot with 7.2 ( $\pm 0.954$ ). The control had the fewest additional exotic species, with an average increase of 1.6 ( $\pm 0.273$ ) species per plot. The thin-only and burn-only treatments were intermediate with 4.2 ( $\pm 0.829$ ) and 3.2 ( $\pm 0.839$ ) additional exotic species per  $1000 m^2$  after treatment, respectively. Undesirable species richness change followed the same pattern, with the greatest increase of 2.8 ( $\pm 0.346$ ) species per  $1000 m^2$  in the thin/burn, and the smallest increase in the control, 0.30 ( $\pm 0.153$ ) species. Again, the thin-only and burn-only treatments were intermediate with an

average 1.93 (+/- 0.176) and 0.8 (+/- 0.153) additional undesirable species per plot, respectively. Overall, 19 of the 26 exotic species found at this level showed the greatest post-treatment increase in frequency in the thin/burn, while only three increased most in the control.

Eleven species, including six undesirable species, were at least marginally significantly different among treatments in the change in plot-level frequency. The undesirable species *Bromus tectorum* was significantly different among treatments ( $p = 0.044$ ), with its largest increase in frequency in the thin/burn with an average increase of 0.133 (+/- 0.088), and the greatest decrease in frequency within the burn-only with an average of -0.033 (+/- 0.088). The three species of thistle all showed significantly different changes in frequency among treatments, with  $p$ -values of 0.042, 0.009, and 0.006 for *Carduus nutans*, *Cirsium arvense*, and *Cirsium vulgare*, respectively. All three thistle species showed the greatest increase in plot-level frequency in the thin/burn, while none increased frequency in the control. *Cynoglossum officinale* was also significantly different among treatments ( $p = 0.041$ ); however, unlike most other exotics, this species showed the greatest increase in frequency in the thin-only with an increased frequency of 0.433 (+/- 0.203) in the thin-only with no change in the control. Finally, *Potentilla recta* was marginally significantly different among treatments ( $p = 0.077$ ). This species also showed the greatest mean increase in frequency in the thin-only (0.367 +/- 0.088) and the least in the control (0.067 +/- 0.033), although it did increase in all treatments.

A number of exotic species not considered undesirable also showed significant treatment differences in plot-level frequency change. *Logfia arvensis* was highly significantly different among treatments ( $p = 0.009$ ). All treatments showed an increase

in frequency for this species. However, the greatest increase was in the thin/burn (0.833 +/- 0.120) followed closely by the burn-only (0.700 +/- 0.173). The control had the smallest increase in frequency of *L. arvensis*, with a gain of only 0.067 (+/- 0.033). *Lactuca serriola* was marginally significantly different among treatments ( $p = 0.076$ ). *L. serriola* increased in both treatments that included with prescribed burning--by 0.433 (+/- 0.219) in the burn-only and 0.267 (+/- 0.088) in the thin/burn. However, the thin-only and control had no change in the frequency of *L. serriola*. *Poa pratensis*, which was also significantly different among treatments ( $p = 0.019$ ), showed the opposite trend, responding to all treatments with thinning. It increased frequency by 0.500 (+/- 0.058) in the thin-only and 0.533 (+/- 0.067) in the thin/burn. *P. pratensis* also increased frequency in the burn-only and control, but to a much smaller extent (0.033 +/- 0.067 and 0.100 +/- 0.058, respectively). *Tragapogon dubious* was the other exotic species with significant ( $p = 0.008$ ) differences among treatments. *T. dubious* increased the most in the thin/burn (0.433 +/- 0.067) and the least in the control (0.067 +/- 0.067), with intermediate increases in the other two treatments.

The average treatment level cover and frequency for undesirable species at the subplot-level (100m<sup>2</sup>) are shown in Table 7. Subplot-level treatment unit averages of overall undesirable cover and richness and exotic richness are also shown, as are results of the BMRPP test of the null hypothesis of no differences among treatments. The eight undesirable species combined varied significantly in cover among treatments ( $p = 0.025$ ), with an overall cover of 1.157 (+/- 0.640) in the thin/burn and a cover of 0.024 (+/- 0.002) in the control. Four of the undesirable species were at least marginally

**Table 6:** Means and standard errors (SE) for treatment (n = 12) change in plot-level exotic frequency with undesirable species in **bold**. Blocked Multi-Response Permutation Procedures (BMRPP) were used to test the null hypothesis of no differences among treatments. A is the chance corrected within group agreement. \* indicates marginally significant (p < 0.1) and \*\* significant (p < 0.05).

Treatment	thin-only		burn-only		thin/burn		control		BMRPP results	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	A	p
<i>Agrostis gigantea</i>	0.2667	0.267	-0.067	0.203	0.4	0.306	0.0667	0.088	0.183	0.103
<i>Apera interrupta</i>	0.1	0.100	0.0333	0.033	0.0667	0.067	0	0.000	-0.116	0.763
<b><i>Bromus tectorum</i></b>	<b>0.067</b>	<b>0.067</b>	<b>-0.033</b>	<b>0.088</b>	<b>0.133</b>	<b>0.088</b>	<b>0.100</b>	<b>0.100</b>	<b>0.238</b>	<b>0.044**</b>
<b><i>Carduus nutans</i></b>	<b>0.033</b>	<b>0.033</b>	<b>0.033</b>	<b>0.033</b>	<b>0.533</b>	<b>0.233</b>	<b>0.000</b>	<b>0.000</b>	<b>0.333</b>	<b>0.042**</b>
<b><i>Centaurea biebersteinii</i></b>	<b>0.200</b>	<b>0.058</b>	<b>0.233</b>	<b>0.088</b>	<b>0.133</b>	<b>0.120</b>	<b>0.167</b>	<b>0.067</b>	<b>-0.205</b>	<b>0.960</b>
<i>Cerastium fontanum</i>	0.033	0.033	0.000	0.000	0.000	0.000	0.033	0.033	-0.143	0.677
<b><i>Cirsium arvense</i></b>	<b>0.133</b>	<b>0.088</b>	<b>0.033</b>	<b>0.033</b>	<b>0.367</b>	<b>0.067</b>	<b>0.000</b>	<b>0.000</b>	<b>0.576</b>	<b>0.009**</b>
<b><i>Cirsium vulgare</i></b>	<b>0.333</b>	<b>0.133</b>	<b>0.133</b>	<b>0.088</b>	<b>0.600</b>	<b>0.173</b>	<b>0.000</b>	<b>0.000</b>	<b>0.485</b>	<b>0.006**</b>
<b><i>Cynoglossum officinale</i></b>	<b>0.433</b>	<b>0.203</b>	<b>0.033</b>	<b>0.033</b>	<b>0.233</b>	<b>0.233</b>	<b>0.000</b>	<b>0.000</b>	<b>0.212</b>	<b>0.041**</b>
<i>Logfia arvensis</i>	0.200	0.200	0.700	0.173	0.833	0.120	0.067	0.033	0.502	0.009**
<i>Lactuca serriola</i>	0.000	0.000	0.433	0.219	0.267	0.088	0.000	0.000	0.267	0.076*
<i>Silene latifolia</i>	0.000	0.000	0.033	0.033	0.000	0.000	0.000	0.000	NA <sup>a</sup>	NA <sup>a</sup>
<i>Melilotus officinalis</i>	0.000	0.000	0.000	0.000	0.033	0.033	0.000	0.000	NA <sup>a</sup>	NA <sup>a</sup>
<i>Myosotis stricta</i>	0.100	0.100	0.033	0.033	0.200	0.100	0.000	0.000	0.059	0.301
<i>Phleum pratense</i>	0.233	0.067	0.233	0.120	0.267	0.088	0.300	0.252	-0.063	0.769
<i>Poa compressa</i>	0.233	0.186	0.233	0.120	0.367	0.186	0.400	0.200	-0.035	0.580
<i>Poa pratensis</i>	0.500	0.058	0.033	0.067	0.533	0.067	0.100	0.058	0.526	0.019**
<i>Polygonum aviculare</i>	0.000	0.000	0.000	0.000	0.167	0.120	0.000	0.000	0.158	0.062*
<b><i>Potentilla recta</i></b>	<b>0.367</b>	<b>0.088</b>	<b>0.133</b>	<b>0.067</b>	<b>0.200</b>	<b>0.058</b>	<b>0.067</b>	<b>0.033</b>	<b>0.314</b>	<b>0.077*</b>
<i>Rumex acetosella</i>	0.067	0.067	0.033	0.033	0.233	0.186	0.033	0.033	-0.030	0.603
<i>Rumex crispus</i>	0.000	0.000	0.000	0.000	0.033	0.033	0.000	0.000	NA <sup>a</sup>	NA <sup>a</sup>
<i>Taraxacum officinale</i>	0.133	0.067	0.267	0.176	0.367	0.219	0.233	0.133	-0.013	0.603
<i>Thlaspi arvense</i>	0.000	0.000	0.033	0.033	0.033	0.033	0.000	0.000	NA <sup>a</sup>	NA <sup>a</sup>
<i>Tragopogon dubius</i>	0.267	0.067	0.167	0.033	0.433	0.067	0.067	0.067	0.409	0.008**
<i>Trifolium species</i>	0.100	0.000	0.233	0.120	0.167	0.088	0.033	0.033	0.133	0.197
<b><i>Verbascum thapsus</i></b>	<b>0.367</b>	<b>0.203</b>	<b>0.233</b>	<b>0.186</b>	<b>0.600</b>	<b>0.000</b>	<b>-0.033</b>	<b>0.033</b>	<b>0.219</b>	<b>0.172</b>
Overall Exotic Richness	4.167	0.829	3.200	0.839	7.200	0.954	1.633	0.273	0.674	0.003**
<b>Overall Undesirable Richness</b>	<b>1.933</b>	<b>0.176</b>	<b>0.800</b>	<b>0.153</b>	<b>2.800</b>	<b>0.346</b>	<b>0.300</b>	<b>0.153</b>	<b>0.754</b>	<b>0.002**</b>

a The skewness of delta approaches infinity, so calculations cannot be done. These species were found in no more than two plots out of a possible 120.

significantly different among treatments. *Carduus nutans* ( $p = 0.024$ ) and *Cirsium vulgare* ( $p = 0.044$ ) are thistle species that showed the greatest cover in the thin/burn (0.196 +/- 0.179 and 0.371 +/- 0.196 percent, respectively). Both species also showed the least cover in the control (0.001 +/- 0.001 and zero percent, respectively). *Cynoglossum officinale* showed marginally significant differences among treatments ( $p = 0.056$ ) with the highest cover (0.031 +/- 0.020 percent) in the thin-only and zero cover in the control. *Verbascum thapsus* was the only other undesirable weed to vary significantly in cover among treatments ( $p = 0.032$ ). It had the highest cover in the thin/burn (0.408 +/- 0.177) and no cover in the control.

Frequency and richness data at the subplot-level show the same trends as the coverage data. Overall undesirable and exotic species richness varied significantly among treatments ( $p = 0.014$  and  $p = 0.011$ , respectively). The treatment with the highest undesirable and exotic richness at the subplot-level is again the thin/burn, with an average 1.237 (+/- 0.538) undesirable species occurrences per 100m<sup>2</sup> and 4.467 (+/- 1.458) exotic species occurrences. The control had the fewest occurrences, with averages of 0.120 (+/- 0.010) undesirable species and 1.367 (+/- 0.366) exotic species per subplot. *Carduus nutans* ( $p = 0.086$ ) and *Cirsium vulgare* ( $p = 0.013$ ) again varied among treatments, with the highest frequencies in the thin/burn (0.147 +/- 0.104 and 0.297 +/- 0.102, respectively) and the lowest in the control (0.007 +/- 0.007 and 0.000, respectively). *Cynoglossum officinale* had the highest frequency in the thin-only (0.073 +/- 0.039), but was not found in the control. The differences were significant ( $p = 0.033$ ) among treatments. *Verbascum thapsus* was highly significantly different ( $p = 0.009$ ) among

treatments, with a maximum frequency of 0.303 (+/- 0.123) in the thin/burn and 0.000 in the control.

### Scale

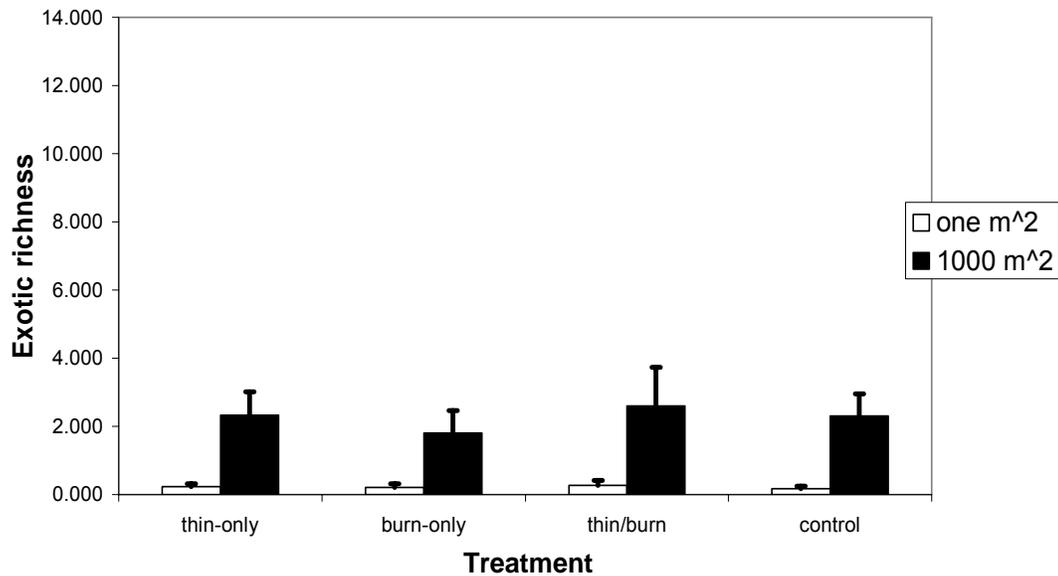
Pre-treatment exotic richness at two scales (1 m<sup>2</sup> and 1000 m<sup>2</sup>) is shown in Figure 6. Figure 7 shows post-treatment exotic richness at these two scales, plus the 100 m<sup>2</sup> scale. The change in exotic richness from pre-to post-treatment varied significantly by treatment at both the quadrat-and plot-levels of analysis. Subplot-level exotic richness was only collected post-treatment in 2003, and was significantly different among treatments that year. Patterns of richness and overall cover for both undesirable and exotic species as groups were similar across all scales. The thin/burn consistently had the highest level of exotic and undesirable species abundance. While the thin-only was generally second to the thin/burn in exotic and undesirable richness and cover at the subplot-level, the cover decreased at the quadrat level, primarily due to a large decrease in the cover of *Centaurea biebersteinii* after treatment.

While results for overall richness and cover were consistent across scales, the frequency and cover of individual species were more variable. Tests for differences in cover among treatments generally yielded similar results at the two scales tested. Only the eight undesirable species were tested at both scales, and with the exception of *Cynoglossum officinale* and *Cirsium arvense*, the results were similar. *Cynoglossum officinale* was only significantly different in cover at the larger subplot scale, while *Cirsium arvense* was only significantly different in cover at the smaller quadrat (1m<sup>2</sup>) scale.

**Table 7:** Means and standard errors (SE) for treatment (n = 12) cover and frequency of undesirable species. Blocked Multi-Response Permutation Procedures (BMRPP) were used to test the null hypothesis of no differences among treatments. A is the chance corrected within group agreement. \* indicates marginally significant (p< 0.1) and \*\*significant (p< 0.05).

Treatment	thin-only		burn-only		thin/burn		control		BMRPP results	
	mean	SE	mean	SE	mean	SE	mean	SE	A	p
<b>cover</b>										
<i>Bromus tectorum</i>	0.001	0.001	0.001	0.001	0.005	0.005	0.004	0.003	-0.116	0.677
<i>Carduus nutans</i>	0.002	0.001	0.001	0.001	0.196	0.179	0.001	0.001	0.043	0.024**
<i>Centaurea biebersteinii</i>	0.097	0.084	0.050	0.045	0.041	0.018	0.006	0.002	-0.013	0.606
<i>Cirsium arvense</i>	0.011	0.010	0.005	0.004	0.062	0.050	0.001	0.001	0.007	0.373
<i>Cirsium vulgare</i>	0.021	0.015	0.006	0.005	0.371	0.196	0.000	0.000	0.285	0.044**
<i>Cynoglossum officinale</i>	0.031	0.020	0.001	0.001	0.015	0.014	0.000	0.000	0.122	0.056*
<i>Potentilla recta</i>	0.021	0.010	0.056	0.055	0.059	0.056	0.011	0.005	-0.028	0.882
<i>Verbascum thapsus</i>	0.019	0.008	0.003	0.002	0.408	0.177	0.000	0.000	0.415	0.032**
All undesirable cover	0.203	0.090	0.122	0.095	1.157	0.640	0.024	0.002	0.285	0.025**
<b>frequency</b>										
<i>Bromus tectorum</i>	0.003	0.003	0.003	0.003	0.027	0.027	0.020	0.015	-0.166	0.677
<i>Carduus nutans</i>	0.010	0.006	0.007	0.007	0.147	0.104	0.007	0.007	0.096	0.086*
<i>Centaurea biebersteinii</i>	0.163	0.099	0.143	0.118	0.163	0.064	0.030	0.010	0.007	0.426
<i>Cirsium arvense</i>	0.030	0.025	0.023	0.019	0.100	0.061	0.007	0.007	0.033	0.162
<i>Cirsium vulgare</i>	0.053	0.023	0.030	0.025	0.297	0.102	0.000	0.000	0.457	0.013**
<i>Cynoglossum officinale</i>	0.073	0.039	0.003	0.003	0.047	0.042	0.000	0.000	0.177	0.033**
<i>Potentilla recta</i>	0.107	0.052	0.187	0.182	0.153	0.139	0.057	0.027	-0.055	0.882
<i>Verbascum thapsus</i>	0.093	0.041	0.013	0.009	0.303	0.123	0.000	0.000	0.387	0.009**
Undesirable richness	0.533	0.163	0.410	0.275	1.237	0.538	0.120	0.010	0.317	0.014**
Exotic richness	2.300	0.611	1.900	0.626	4.467	1.458	1.367	0.366	0.399	0.011**

**Figure 6:** Pre-treatment exotic richness at the quadrat (1 m<sup>2</sup>) and plot (1000 m<sup>2</sup>) level. Error bars are treatment means plus one standard error.

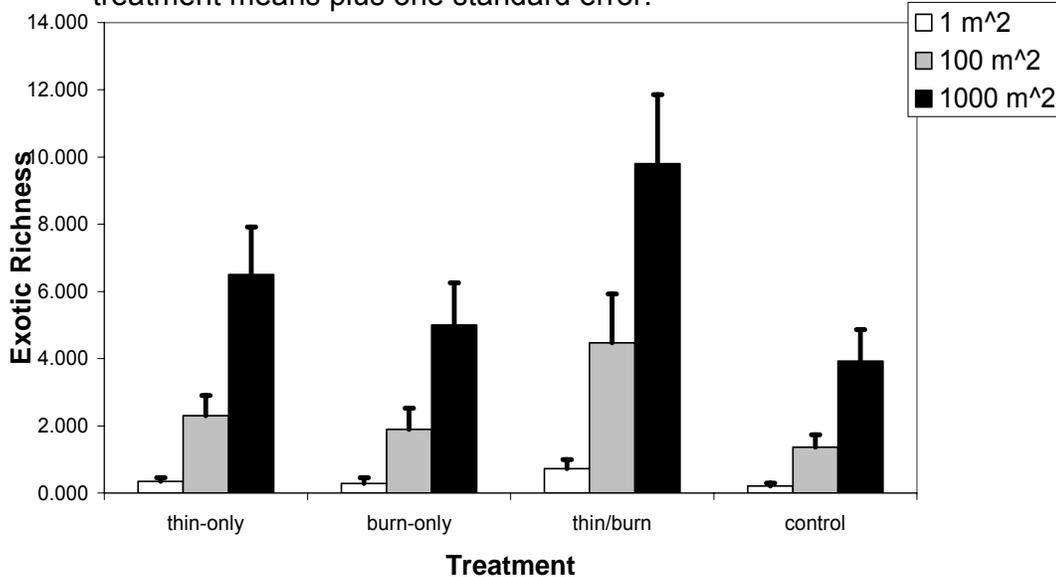


Differences in frequency for individual species were more pronounced among scales. At the quadrat-level, only six species (three undesirable) were significantly different among treatments, while at the plot level 11 species (six undesirable) showed significant changes in frequency after treatment. Undesirable species that were different at the plot-level in frequency change, but not at the quadrat-level, include: *Bromus tectorum*, *Cirsium arvense*, *Cynoglossum officinale*, and *Potentilla recta*. The quadrat-level had one undesirable species, *Verbascum thapsus*, with significantly different change in frequency among treatments that was not significant at the plot level.

Other non-undesirable exotic species that were different at the plot-level but not at the quadrat-level included: *Lactuca serriola*, *Polygnum aviculare*, and *Tragapogon dubious*. However, *Taraxacum officinale* is only different at the quadrat-level and not the plot-level. Subplot-level undesirable weed frequency tests for differences among

treatments reveal the same results as the quadrat level, except that *Cynoglossum officinale* was also significantly different among treatments.

**Figure 7:** Post-treatment exotic richness at the quadrat (1 m<sup>2</sup>), subplot (100 m<sup>2</sup>), and plot (1000 m<sup>2</sup>) levels. Error bars are treatment means plus one standard error.



### Indicator Species Analysis

Pre-and post-treatment relationships of exotic species with treatment are shown in Table 8. No exotic species were significantly ( $IV > 20$  and  $p < 0.05$ ) associated with any particular treatment more strongly than the control before treatment. However, in the post-treatment data a different pattern emerges. All of the treatments had at least one exotic species as a significant indicator. *Poa pratensis* was a significant indicator of the thin-only ( $IV = 30.9$ ,  $p = 0.039$ ); *Logfia arvensis* was a significant indicator of the burn-only ( $IV = 23.3$ ,  $p = 0.011$ ); and six exotic species were indicators of the thin/burn including the two species that were indicators of the thin-only and burn-only. Also, two thin/burn indicators are undesirable species (*Verbascum thapsus*  $IV = 53.3$ ,  $p = 0.001$  and

*Cirsium vulgare* IV = 20, p = 0.021), and two others--*Trifolium* species (IV = 40.1, p = 0.018) and *Taraxacum officinale* (IV = 50.4, p = 0.023)--are widespread exotics in this region (Weaver et al. 2001).

### Covariate Analysis

The model with only treatment and block as explanatory variables, and change in exotic richness as the response variable, showed no evidence of violation of assumptions as the p-value for Levene's test was 0.262. The final model is displayed in Table 9. Block and treatment were both significant ( $p < 0.001$ , extra sums of squares F-test) in predicting pre-to post-treatment change in exotic species richness, and explained 61.3 percent of the variation ( $R^2 = 0.613$ ). However, there was no evidence that the effect of treatment varied by block, as the interaction term was not significant based on the extra sums of squares F-test ( $p = 0.121$ ). Therefore, the interaction term was removed from the model. Block three was aliased so that the other blocks could be compared with it. Both blocks one and two varied significantly from block three, although in opposite directions (block one had the least change in exotic species richness, while block two had the most). The control "level" of treatment was also aliased so that the other treatments could be compared pair-wise with it for change in exotic richness. At the plot level, all of the treatments were highly significantly different ( $p < 0.002$ ) than the control. All showed increases in exotic richness compared to the control, with an estimated increase beyond that of the control of 1.57 species per plot in the burn-only, 2.53 in the thin-only, and 5.57 in the thin/burn. None of the 95 percent confidence intervals included zero.

**Table 8:** Results of pair-wise Indicator Species Analyses between each treatment and the control for exotic species (n = 120). The treatment column shows which treatment a species was grouped into, IV is the indicator value, and p is the probability of obtaining such a high indicator value with the data randomized between treatments. \* indicates significant ( $p < 0.05$ ).

<b>thin-only vs control</b>						
<b>Species</b>	<b>Pre-Treatment</b>			<b>Post-Treatment</b>		
	<b>Treatment</b>	<b>IV</b>	<b>p</b>	<b>Treatment</b>	<b>IV</b>	<b>p</b>
<i>Agrostis gigantea</i>	control	3.3	1	thin-only	6.7	0.511
<i>Apera interrupta</i>	NA*	0	NA	thin-only	3.3	1
<i>Bromus tectorum</i>	control	6.7	0.483	NA	0	NA
<i>Centaurea biebersteinii</i>	thin-only	16.5	0.242	thin-only	17.4	0.161
<i>Cirsium arvense</i>	thin-only	3.2	1	thin-only	3.1	1
<i>Cirsium vulgare</i>	thin-only	3.3	1	thin-only	3.3	1
<i>Cynoglossum officinale</i>	NA	0	NA	thin-only	3.3	1
<i>Myosotis stricta</i>	NA	0	NA	thin-only	3.3	1
<i>Phluem pratense</i>	thin-only	5.7	0.606	control	11.1	0.379
<i>Poa compressa</i>	thin-only	3.3	1	thin-only	4.4	1
<i>Poa pratensis</i>	thin-only	4.7	1	thin-only	30.9	0.039*
<i>Potentilla recta</i>	thin-only	12.7	0.724	thin-only	13.3	0.811
<i>Taraxacum officinale</i>	thin-only	43.4	0.082	thin-only	31.2	0.575
<i>Tragapogon dubius</i>	control	7.8	0.681	control	4.7	1
<i>Trifolium Spp.</i>	thin-only	15.2	0.442	thin-only	17.4	0.381
<i>Verbascum thapsus</i>	thin-only	3.3	1	thin-only	10	0.241
<b>burn-only vs control</b>						
<b>Species</b>	<b>Pre-Treatment</b>			<b>Post-Treatment</b>		
	<b>Treatment</b>	<b>IV</b>	<b>p</b>	<b>Treatment</b>	<b>IV</b>	<b>p</b>
<i>Agrostis gigantea</i>	burn-only	11.7	0.311	burn-only	10	0.234
<i>Bromus tectorum</i>	control	4.4	1	NA	0	NA
<i>Centaurea biebersteinii</i>	burn-only	12.1	0.262	burn-only	16	0.131
<i>Cirsium arvense</i>	control	3.3	1	control	3.3	1
<i>Lactuca serriola</i>	NA	0	NA	burn-only	6.7	0.463
<i>Logfia arvensis</i>	NA	0	NA	burn-only	23.3	0.011*
<i>Myosotis stricta</i>	NA	0	NA	burn-only	3.3	1
<i>Phluem pratense</i>	control	6.7	0.513	control	14.8	0.145
<i>Poa compressa</i>	NA	0	NA	burn-only	1.7	1
<i>Poa pratensis</i>	burn-only	8.5	0.982	burn-only	8.6	0.716
<i>Potentilla recta</i>	burn-only	15.4	0.505	burn-only	21.8	0.219
<i>Taraxacum officinale</i>	burn-only	23.5	0.805	control	23.7	0.957
<i>Tragapogon dubius</i>	control	8.8	0.489	control	4.7	1
<i>Trifolium Spp.</i>	control	15.7	0.145	control	13.3	0.378
<i>Verbascum thapsus</i>	NA	0	NA	burn-only	3.3	1

Table 8 continued

Species	thin/burn vs control					
	Pre-Treatment			Post-Treatment		
	Treatment	IV	p	Treatment	IV	p
<i>Agrostis gigantea</i>	control	3.3	1	thin/burn	10	0.249
<i>Apera interrupta</i>	thin/burn	3.3	1	thin/burn	6.7	0.476
<i>Bromus tectorum</i>	control	4.4	1	thin/burn	3.3	1
<i>Carduus nutans</i>	NA	0	NA	thin/burn	13.3	0.11
<i>Centaurea biebersteinii</i>	thin/burn	10	0.385	thin/burn	11.1	0.371
<i>Cirsium arvense</i>	control	3.3	1	thin/burn	9.7	0.346
<i>Cirsium vulgare</i>	thin/burn	3.3	1	thin/burn	20	0.021*
<i>Logfia arvensis</i>	NA	0	NA	thin/burn	26.7	0.005*
<i>Myosotis stricta</i>	NA	0	NA	thin/burn	16.7	0.052
<i>Phleum pratense</i>	thin/burn	12.7	0.23	thin/burn	13.5	0.629
<i>Poa compressa</i>	NA	0	NA	thin/burn	10.7	0.354
<i>Poa pratensis</i>	control	10	0.786	thin/burn	43.2	0.001*
<i>Polygonum aviculare</i>	NA	0	NA	thin/burn	3.3	1
<i>Potentilla recta</i>	thin/burn	12.2	0.618	thin/burn	20.9	0.254
<i>Rumex acetosella</i>	NA	0	NA	thin/burn	6.7	0.518
<i>Taraxacum officinale</i>	thin/burn	37.2	0.166	thin/burn	50.4	0.023*
<i>Tragapogon dubius</i>	control	10	0.224	thin/burn	5.6	0.98
<i>Trifolium</i> spp.	thin/burn	24.2	0.163	thin/burn	40.1	0.018*
<i>Verbascum thapsus</i>	thin/burn	3.3	1	thin/burn	53.3	0.001*

NA\*--species were not found in one sampling year but were in the other.

**Table 9:** Parameter estimates for the regression of change in plot-level (1000 m<sup>2</sup>) exotic richness with block and treatment as explanatory variables (n = 120 plots).

Parameter	$\beta$	Significance	95% Confidence Interval	
			Upper Bound	Lower Bound
intercept	1.76	0.000	0.95	2.57
block 1	-1.35	0.001	-2.16	-0.54
block 2	0.98	0.019	0.16	1.79
block 3	0.00	.	.	.
thin-only	2.53	0.000	1.60	3.47
burn-only	1.57	0.001	0.63	2.50
thin/burn	5.57	0.000	4.63	6.50
control	0.00	.	.	.

R Squared = 0.613 (Adjusted R Squared = 0.596)

The parsimonious model with environmental variables and block explained 62.3 percent of the variation in the change of exotic species plot level richness from pre-to

post-treatment (Table 10). Levene’s test provided no evidence of heteroscedacity ( $p = 0.662$ ). The final parsimonious model had only five explanatory variables, four environmental variables and block. The four significant ( $p < 0.015$ ) environmental variables are canopy completeness, change in native richness, maximum bole charring, and the proportional change in sapling density. Parameter estimates predict that a 5.6 percent increase in canopy completeness on a plot results in one fewer new exotic species (95 percent confidence interval from 0.34 to 1.69 species). With the addition of 10 native species, one additional exotic species is expected (95 percent confidence interval from 0.2 to 1.7). A 50 percent reduction in the sapling density results in an estimated increase of one exotic species (95 percent confidence interval from 0.46 to 1.55). Finally, an additional 2.38 m of bole scarring due to burning results in an estimated additional exotic species, with a 95 percent confidence interval from 0.31 to 1.69.

**Table 10:** Final parsimonious model with change in plot-level (1000 m<sup>2</sup>) exotic richness as the response variable and environmental factors as the explanatory variables (n = 120 plots).

Parameter	$\beta$	Significance	95% Confidence Interval	
			Lower Bound	Upper Bound
Intercept	2.76	0.025	0.36	5.17
block 1	-0.94	0.030	-1.78	-0.09
block 2	1.19	0.005	0.36	2.01
block 3	0.00 <sup>a</sup>	.	.	.
Canopy completeness	-0.18	0.004	-0.30	-0.06
Change in native richness	0.10	0.012	0.02	0.17
Maximum bole char	0.42	0.005	0.13	0.71
Change in sapling density	2.01	0.000	0.93	3.09

a This parameter is set to zero because it is redundant.  
R Squared = 0.623 (Adjusted R Squared = 0.603)

Results of the tests for treatment differences in environmental variables that were significantly correlated with plot-level exotic richness change are shown in Table 11. Additionally, treatment means and standard errors are shown for each treatment (n = 12).

There was no evidence of unequal variance for the proportional change in sapling density, as Levene's test was not significant ( $p > 0.1$ ). Normal p-p plots revealed no extreme deviation from normality. Block was not significant ( $p > 0.1$ ) and was eliminated from the model. The control had an increase in sapling density from pre-to post-treatment, with an average increase of 18.2 percent (standard error +/- 6.5percent). This was significantly ( $p < 0.05$ ) different than all of the other treatments which had decreases in sapling density on average. The thin/burn had the largest average decrease in sapling density, 85.6 percent (+/- 6.5 percent). This was also significantly ( $p < 0.05$ ) different than each of the other treatments. The thin-only and burn-only also showed decreases in sapling density that were significantly ( $p < 0.05$ ) different than the thin/burn and control, but not from each other.

Levene's test for canopy completeness was not significant ( $p > 0.1$ ), and normal p-p plots showed no serious deviations from the normal distribution. Block was not significant ( $p > 0.1$ ) and was eliminated from the model. The thin/burn and thin-only had the most open canopies with an average of 36 percent (standard error +/- 2.6 percent) and 44 percent (+/- 3.1 percent) of the sample points with live canopy cover, respectively. They were not significantly different from each other ( $p > 0.05$ ), but each was significantly ( $p < 0.05$ ) different than the burn-only and control. The burn-only and control were not significantly ( $p > 0.1$ ) different from each other--with an average of 65.3 percent (+/- 3.2 percent) and 70 percent (+/- 2.5 percent) of sampling points with live canopy cover, respectively.

The height of bole char was not significantly different among blocks ( $p > 0.1$ ), so

block was eliminated from the model. Levene's test and normal p-p plots revealed no violations of assumptions. Bole char was only recorded in the two burned treatments. However, there were significant ( $p < 0.05$ ) differences among the two burned treatments. The thin/burn had an average bole char height of 3.3 m (standard error +/- 0.53 m) while the burn-only had an average of 1.1 m (+/- 0.24 m).

Native species richness increased in all treatments from pre-to post-treatment. However, in this case block was significant and therefore retained in the model. This left too few degrees of freedom for Levene's test to be performed; however, plots of predicted versus residual values revealed no strong evidence of unequal variance, and normal p-p plots showed no obvious deviation from normality. The control (mean increase of 15.97 species, standard error +/- 0.09) and the burn-only (16.13 +/- 0.81) were not significantly different from each other ( $p > 0.1$ ) but each was significantly different from both the thin/burn and thin-only ( $p < 0.05$ ). The thin/burn (20.90 +/- 1.79) and the thin-only (19.96 +/- 1.00) had the largest increases in native richness, although they were not significantly different from each other.

**Table 11:** Treatment means and standard errors for environmental variables significantly correlated with plot exotic richness change. Least significant difference (LSD) tests were used to test for significant treatment differences (n = 12).

<b>Treatment</b>	<b>proportional sapling change (SE)</b>	<b>canopy completeness (SE)</b>	<b>meters bole char (SE)</b>	<b>change in native richness (SE)</b>
thin-only	0.200 (0.026) <sup>a</sup>	0.440 (0.031) <sup>a</sup>	0	19.96 (1.00) <sup>a</sup>
burn-only	0.334 (0.064) <sup>a</sup>	0.653 (0.032) <sup>b</sup>	1.140 (0.240) <sup>a</sup>	16.13 (0.81) <sup>b</sup>
thin/burn	0.856 (0.065) <sup>b</sup>	0.360 (0.026) <sup>a</sup>	3.267 (0.530) <sup>b</sup>	20.90 (1.79) <sup>a</sup>
control	-0.182 (0.065) <sup>c</sup>	0.700 (0.025) <sup>b</sup>	0	15.97 (0.09) <sup>b</sup>

<sup>a</sup>Different letters indicate significant treatment differences at  $\alpha = 0.05$

## Discussion

### Differences among treatments

The results of this study are consistent with one of the dominant paradigms in invasion biology: disturbance facilitates invasion. When all exotic and undesirable species were grouped together for analysis, their responses varied significantly among treatments. These differences were evident at all scales of analysis. The control, which had no disturbance due to treatment, generally had the lowest exotic abundance, except in quadrat-level (1 m<sup>2</sup>) cover change.

The thin/burn consistently had the greatest abundance of both exotic and undesirable species. This pattern was consistent across abundance measures and scales of analysis (see Figures 6 and 7). At each level of cover analyzed (1 m<sup>2</sup> and 100 m<sup>2</sup>) the thin/burn had almost an order of magnitude higher cover of undesirable and exotic species than the other treatments. Richness showed a similar pattern, although differences were not as pronounced.

Other analyses provide additional evidence that the thin/burn was different from the other treatments. The NMS demonstrates that the exotic plant community in several of the thin/burn plots is very different from any of the other plots in the study, and more variable. Cover values from each of the 21 exotic species are used to derive distances among plots. This would indicate that the thin/burn has several species with cover values that are very different from other treatments. The plot of the exotic species in plot space (Figure 5) suggests that several species (*Cirsium arvense*, *Cirsium vulgare*, *Carduus nutans*, *Trifolium* species, and *Verbascum thapsus*) showing evidence of differential treatment responses may be important in determining the position of some thin/burn

plots. Finally, the Indicator Species Analysis provides further evidence that the thin/burn had the most exotic species with strong response.

Generally the thin-only had the second highest levels of exotic abundance. However, the thin-only showed a decrease in quadrat-level cover after treatment. This was the only instance where exotic species as a group declined from pre-to post-treatment. This contrasts with the richness data where the thin-only increased in overall exotic richness at both the quadrat and plot levels. At both these levels only the thin/burn showed a larger increase in exotic richness than the thin-only. At the subplot level (which was not measured before treatment), the thin-only had the second highest cover and richness for undesirable species, with the thin/burn again having the most. Except in quadrat-level cover, the burn-only was always intermediate between the control and other treatments in exotic abundance. While the thin-only and burn-only generally showed increases in exotic richness and cover greater than that of the control, adding together the effects of each treatment does not explain all of the invasion observed in the thin/burn, suggesting a synergistic relationship.

*Centaurea biebersteinii* was primarily responsible for the decrease in quadrat-level cover in the thin-only. While the cover for this species decreased--frequency increased. This could be because *C. biebersteinii* does not grow well in slash piles, yet the thinning that creates slash also creates microsites where additional plants could establish. An alternative explanation is that this is an artifact of changing cover estimates from classes in the pre-treatment sampling to actual percent cover post-treatment

Several studies have found that higher levels of exotic invasion accompany increased levels of disturbance (Griffis et al. 2001; Crawford et al. 2001; Watson and

Renney 1974). More intense treatments would be expected to reduce sapling density, decrease canopy completeness, and result in increased bole charring. In this study, all three of these environmental variables were correlated with exotic richness change such that the more intense the disturbance (greater proportional decrease in saplings, less canopy completeness and more bole charring) the greater the predicted increase in exotic richness. Bole char and the proportional change in sapling density were significantly ( $p < 0.05$ ) different in the thin/burn than the other treatments. Although the thin/burn was not significantly different than the thin-only in canopy completeness, it was significantly different than the other treatments (burn-only and control). Therefore, the increased exotic invasion observed in the thin/burn may be related to increased disturbance intensity.

Additionally, the thin/burn was disturbed more frequently, by two separate disturbance agents in two consecutive years. The plant community can be altered in different ways depending on the frequency of disturbance (Tveten and Fonda 1999; Hobbs and Huenneke 1992). In inland Northwest ponderosa pine forests, fire was the dominant disturbance under which understory species evolved, with an average fire-return interval of 3-38 years (Hessburg and Agee 2003). Therefore, many native plants may not be adapted to more frequent disturbances, and exotics may gain a competitive advantage. Indeed some studies indicate that exotic plants may need frequent disturbance to maintain dominance (Gross and Werner 1982; Petryna et al. 2002).

An alternative explanation could be provided by the fluctuating resource hypothesis (Davis et al. 2000). Burning may release nutrients and stimulate nutrient cycling. Removing trees by thinning may increase the available resources by decreasing

competition from overstory trees. In fact, understory productivity in ponderosa pine forests has been shown to be limited by competition from trees for soil nutrients and water, not light (Riegel et al. 1992). When combined, treatments may reach a threshold of resource availability necessary for exotics to invade or establish. Individually treatments may not be sufficiently intense to reach this threshold. There is evidence to support the idea of disturbances (fire and mechanical cutting) acting in a synergistic fashion to promote invasion (Hobbs and Huenneke 1992).

### Environmental Variables

Over half the variability in plot-level change in exotic species richness was explained by four environmental variables and block. Decreased competition from resident plants after disturbance has been shown to facilitate invasion (Alpert et al. 2000; Weaver et al. 2001; Lonsdale 1999). Canopy completeness and the proportional change in saplings per hectare may also be related to the level of competition from trees. This could be competition for light, although in these forests underground resources may be more important than light levels (Riegel et al. 1992). With decreased competition, additional resources would become available, which may facilitate invasion (Davis et al. 2000; Huenneke et al. 1990; Thompson et al. 2001; Wedin and Tilman 1996).

The change in native richness was positively correlated with the change in exotic richness. This supports the hypothesis that there is no fundamental difference in the way exotic and native species invade (Thompson et al. 2001; Meiners et al. 2004; Levine and D'Antonio 1999). Treatments that create conditions that are favorable for establishment may favor both exotic and native plant invasions, and the first species to get there may successfully establish at the site (Davis 2003). Alternatively, it may be that

environmental conditions that allow increased native richness, such as high resource availability, may also allow increased exotic richness (Stohlgren et al. 2002). In this study, the thin/burn had evidence of the most invasion, but also had significantly ( $p < 0.05$ ) more new native species than all other treatments except the thin-only.

Some environmental factors associated with invasion in other studies were not significant here. Fornwalt et al. (2003) found that topographic position was more important than management history in determining the extent of invasion. However, effective aspect was not important in explaining variation in the change in exotic species richness in this study. This may be because slopes are relatively gentle in the study area, and all blocks are nearly the same elevation. Many opportunistic exotic plant species depend on bare-ground or disturbed soil to establish (Beck 1999; Sheley et al. 1996; Sieg et al. 2003). Disturbance may act to create patches of ground that are favorable to exotic establishment, especially during the early stages of invasion (Masters and Sheley 2001; Hobbs and Huenneke 1992). However, in this study mineral soil was not significant in explaining changes in exotic species richness. It may be that mineral soil is not the limiting factor for exotic establishment at this study site, but something else such as propagule pressure may be.

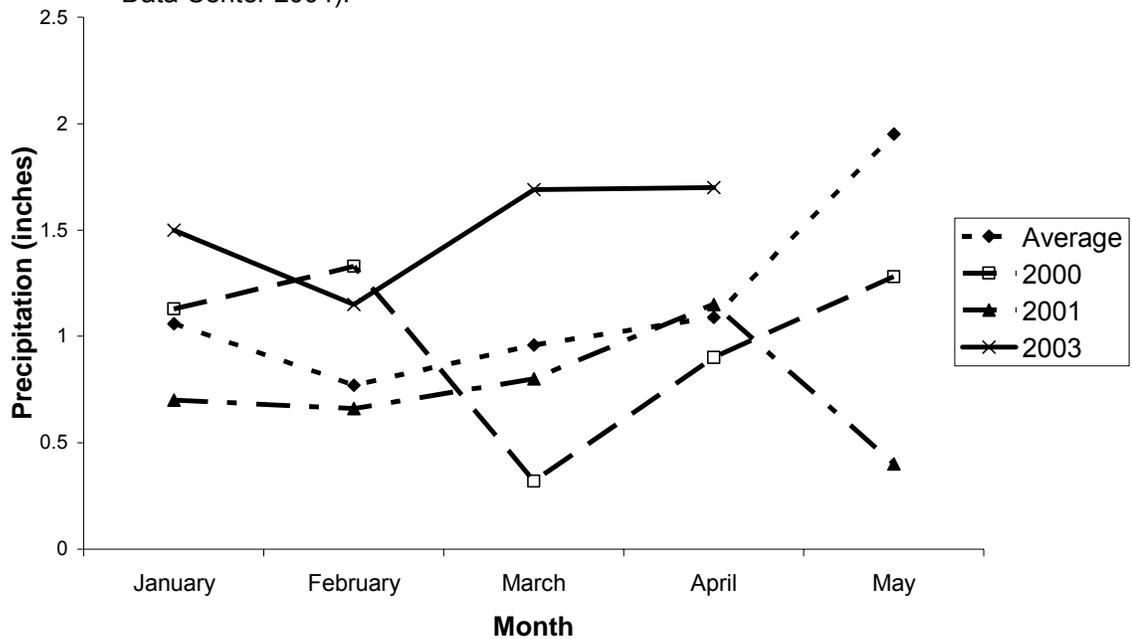
While there is evidence of a positive correlation between invasion by exotic and native species, there is no clear evidence of a pattern of invasion that varies with species richness. Pre-treatment diversity and richness were not significant in explaining variation in the change in exotic richness. Treatment units were chosen to be relatively homogeneous, and blocks were close together, so there may not have been enough variation present before treatments to sort out the relationship between native richness

and exotic invasion. Davis and Pelsor (2001) also found no relationship between native and exotic richness. Alternatively, it may be that diversity itself does not determine invasion success, but factors that covary with diversity do (Meiners et al. 2004; Levine 2000; Levine and D'Antonio 1999; Thompson et al. 2001; Lonsdale 1999).

### Measuring Exotic Abundance

Since early phases of population growth are more easily controlled (Radosevich et al. 2003), it is important to monitor the response of each species to see which may develop into a troublesome species (Sieg et al. 2003). Using both cover and frequency as measures provided multiple ways to detect exotic plant response to treatment. Cover is not dependent on the area sampled, but is variable due to yearly climatic differences. Figure 8 shows that winter/spring precipitation was higher in 2003 than in pre-treatment sampling years. Therefore, cover may be higher for all species (including exotics) in 2003 due only to annual variability. In this study, great care was taken to calibrate all members of the field crew to ensure fidelity in coverage estimates. Additionally, the supervisor of the crew participated in sampling every year, further ensuring that cover values were estimated similarly. While cover values may change due to yearly climatic variation, frequency represents the distribution of individual species, so is dependent on the area measured (McCune and Grace 2002).

**Figure 8:** Precipitation for the first five months of each sampling year along with average monthly values for Missoula, Montana (National Climatic Data Center 2004).



Data for May 2003 were not available.

Results for differences among treatments in cover and frequency were not always consistent at different measurement scales. The large, plot-level (1000 m<sup>2</sup>) scale offered a coarse measurement of changes in species frequency. Common species would be found somewhere on most plots, no matter what the treatment, although their actual abundance varied greatly, depending on the number of occurrences per plot. Therefore, this scale may be too coarse to detect changes in frequency for the more common species, as frequency approaches 100 percent as the scale gets large (McCune and Grace 2002). For example, at every scale other than the plot level, *Verbascum thapsus* was significantly different among treatments in both cover and frequency. This could be because *V. thapsus* occurred often per 1000 m<sup>2</sup> in both the thin/burn and thin-only, and smaller scales were needed to provide a better estimate of changes in frequency.

In contrast, the large scale detected significant differences among treatments for many species that were not significant at the smaller scales (1 m<sup>2</sup>, 100 m<sup>2</sup>). Many of the exotic species were not abundant (all exotics combined covered less than two percent of the thin/burn treatment where their cover was the highest). Rare species may not be present frequently enough at the quadrat level (which samples only 1.2 percent of the plot area) to give a good estimate of cover or frequency. By sampling both cover and frequency at multiple scales, the ability to detect differential performances among individual species was maximized for both common and rare species.

#### Monitoring Individual Undesirable Species

From the Indicator Species Analysis and BMRPP tests it is obvious that some exotic species responded more strongly to certain treatments. Perhaps the most notorious species in the study, *Centaurea biebersteinii*, has a broad ecological amplitude, with the ability to colonize a variety of habitats (Watson and Renney 1974). It has become one of the most prevalent invaders in the West. Present in only 48 counties in the Pacific Northwest in 1980; by 1999 it was found in 326 counties, including all counties of ID, MT, WA, and WY (Sheley et al. 1999). In fact *C. biebersteinii* is the most serious rangeland weed in Montana, infesting more than 4.5 million acres in 1996 (Sheley et al. 1996). This species is known to respond quickly to disturbance, increasing plant density as the intensity of the disturbance increases (Watson and Renney 1974). Moreover, fire may be the type of disturbance that promotes colonization for *C. biebersteinii* (Sheley et al. 1999). Adding nitrogen to a system, which may occur the first year after burning (Deluca and Zouhar 2000), has been shown to shift the competitive advantage to *C. biebersteinii* (Blicker et al. 2002).

In treated areas, light and resource levels may be increased, while competition is decreased. Therefore, *C. biebersteinii* would be expected to respond strongly. However, in this study there was no evidence of differences among treatments in cover or frequency, at any scale of sampling. Although *Centaurea biebersteinii* is a troublesome rangeland invader, shading by overstory trees may prevent invasion in forests (Sheley et al. 1996; Watson and Renney 1974). Weaver et al. (2001) found that *C. biebersteinii* was present at only low levels in the *P. menziesii*/*S. albus* habitat type (Pfister et al. 1977), and did not invade undisturbed vegetation. In the short term, at least, this study indicates that *C. biebersteinii* may not be a disruptive invader after restoration in these ecosystems.

*Potentilla recta* is becoming one of the most prominent invaders in the Northern Rockies, exponentially spreading throughout the region, much like *Centaurea biebersteinii* (Rice 1999). While *C. biebersteinii* is declining on many sites, *P. recta* is increasing (Rice 1999). This species is capable of invading a broad array of ecosystems due to its large ecological amplitude (Rice 1999; Rice 1993). Additionally, prescribed burns increase seedling recruitment in *P. recta* (Lesica and Martin 2003). In the thin-only, *P. recta* was found on 11 plots where it was not found before treatment. In comparison, it spread to only two additional plots in the control. Differences among treatments were marginally significant at the plot level (1000 m<sup>2</sup>), but neither cover nor frequency were different at any other scale. Thus there is only weak evidence of unequal performance among treatments for *P. recta*, in the short term. Overstory shading has also been shown to limit the establishment of *P. recta* (Rice 1999), and might have curtailed aggressive invasion in this study.

Analyzing *Bromus tectorum* reveals a similar pattern. Change in frequency is only significantly different among treatments at the largest scale (1000 m<sup>2</sup>), while cover is never different. *B. tectorum* was found on four additional plots after treatment in the thin/burn (where it increased the most). However, this species was also found on three additional plots in the control, while decreasing in the burn-only, so although the differences are significant, they may not be due to treatment. Another study found dramatic increases in *Bromus tectorum* cover after wildfires in ponderosa pine forests, with 35 percent cover five years after a severe fire and six percent cover five years after a light fire (Johnson 1998). However, in the same study, the cover of *B. tectorum* was no more than one percent the first year after burning (Johnson 1998), indicating that this species may take longer to respond. Alternatively, the seasonality of burning in the Lubrecht study may not have been favorable to *B. tectorum*, as spring burns could kill plants before they were able to set seed.

*Cirsium arvense* is the species most commonly listed as undesirable on state weed lists (Skinner et al. 2000). It is the most widespread invasive thistle, and is considered the most difficult to control due to extensive lateral root systems that can respond to disturbance by forming buds (Morishita 1999). It has been shown to increase in abundance following fire (Scherer et al. 2000; Tu et al. 2000). In this study, detecting differential performance among treatments depended on the scale and the measurement (cover or frequency). In the thin/burn, it was found in an additional 11 plots after treatment where it had not occurred prior to treatment. At the same time, the control showed no change in frequency. This difference was highly significant among treatments ( $p = 0.009$ ). The change in cover at the quadrat level revealed marginally significant

evidence of differences among treatments, while other measures revealed no differences among treatments. The highest subplot level cover was 0.062 percent in the thin/burn, indicating that this species was not very common, in which case large scales may be more sensitive to treatment differences.

*Carduus nutans* is a biennial thistle that has alleopathic effects on other plant species (Beck 1999; Wardle et al. 1993). Seeds are viable for over 15 years, so substantial control efforts may be needed once this species has established (Burnside et al. 1981; Beck 1999). As this species invades an area, it may stimulate recruitment of its own seedlings (Beck 1999; Wardle et al. 1993). In this study there was consistently evidence of treatment differences in the abundance of *C. nutans*. The thin/burn had the greatest abundance, while the control showed no change in either frequency or cover.

*Cirsium vulgare* is another biennial thistle that competes with native plant species, decreasing their productivity (Beck 1999). Additionally, *C. vulgare* has been shown to decrease *Pinus ponderosa* seedling growth (Randall and Rejmánek 1993). *Cirsium vulgare* showed no change in frequency or cover in the control, while consistently having the highest levels of cover and frequency in the thin/burn. These differences were always significant among treatments. In fact, *C. vulgare* was a significant indicator of the thin/burn treatment. Other studies demonstrated that this species responds strongly after fire and other disturbances (Scherer et al. 2000; Beck 1999; Petyrna et al. 2002). However, *C. vulgare* may be replaced by native species within a short time frame if disturbance does not continue (Dejong and Klinkhamer 1988; Petyrna et al. 2002).

*Cynoglossum officinale* is a successful invader because of its prolific seed production, seed dormancy, seed dispersal (by attaching onto animals), and ability to

withstand environmental stresses (Upadhyaya et al. 1988). Biennials, such as *C. officinale*, have been considered part of an opportunistic group of plants that exploits rare conditions (such as those after disturbance) to establish and grow (Dejong and Klinkhamer 1988). *C. officinale* may maintain itself in the community longer, as Dejong and Klinkhamer (1988) found that it disappeared from only seven percent of plots after two years. In this study there was evidence of treatment differences for *Cynoglossum officinale* in both cover and frequency at different scales. Unlike most of the other species, *C. officinale* responded most strongly in the thin-only, with the lowest cover and frequency in the control.

*Verbascum thapsus* has a broad ecological amplitude and may be a troublesome weed in some areas (Parker et al. 2003). It has great seed longevity, so once it invades a site it may be hard to eradicate (Burnside et al. 1996; Kivilaan and Bandurskii 1973; Gross and Werner 1982). *Verbascum thapsus* showed some of the most significant differences among treatments in cover and frequency. It was also a significant indicator species of the thin/burn treatment. Only at the plot level did change in frequency show no evidence of differences among treatments. Neither frequency nor cover of this species changed in the control from pre-to post-treatment, at any scale. This corresponds with Fornwalt et al.'s (2003) findings that *Verbascum thapsus* was the only species to have significantly more cover on managed sites than protected sites. However, longer term impacts of *Verbascum thapsus* invasion are uncertain, as it has been shown to give way to native plants within ten years (Gross and Werner 1982). *V. thapsus* depends on continual recolonization to remain in a community. Recolonization appears to depend on bare-

ground, which is often ephemeral (Gross and Werner 1982). Therefore, this species may not be a persistent problem after restoration.

#### Other Notable Exotic Species

Grasses are often spread intentionally by humans, facilitating invasions (D'Antonio and Vitousek 1992). D'Antonio and Vitousek (1992) argue that exotic grasses compete effectively with native grasses, respond strongly after fire, and have the potential to alter ecosystem properties. Indeed, *Poa pratensis* has replaced native communities in grasslands throughout the world (Mack et al. 2000; Daubenmire 1978; Baker 1986). In this study *Poa pratensis* was an indicator of both the thin/burn and thin-only treatments suggesting that it responded strongly to thinning. Additionally, there was evidence that both frequency and cover of *Poa pratensis* varied among treatments at all scales. Weaver et al. (2001) found *Poa pratensis* was widespread in the Northern Rockies and capable of invading undisturbed sites in the *Psuedotsuga menziesii*/*Symphoricarpus albus* habitat type (Pfister et al. 1977).

*Taraxacum officinale* and *Trifolium* species were both significant indicators of the thin/burn treatment. Weaver et al. (2001) found that both of these species are widespread in the Northern Rockies and likely to establish high coverage's and invade undisturbed sites in the *P. menziesii*/*S. albus* habitat type (Pfister et al. 1977). However, the changes in cover or frequency of *T. officinale* and *Trifolium* species were each marginally significant at only one combination of scale and abundance measure. Pre-treatment, neither of these species was a significant indicator of the thin/burn, but there is some evidence that there was more of each in the thin/burn (the indicator values are large but not significant). Therefore, the change these species showed from pre-to post-treatment

may not be as significant because there was more in the thin/burn to begin with. In the United States these species are not a high priority for control (Reinhart et al. 2001), but *Taraxacum officinale* is an undesirable weed in Saskatchewan and Quebec (Steward-Wade et al. 2002), and *Trifolium* species are successful invaders that compete well as weeds (Baker 1986).

*Logfia arvensis* was significantly different among treatments at all scales and measures. This species responded strongly to burning, increasing cover and frequency in both the thin/burn and burn-only. Despite this strong response, *Logfia arvensis* may not be a threatening invader. There is little literature documenting this species' long-term invasion success.

Change in plot-level frequency also differed significantly among treatments for *Tragapogon dubious*. Like most other species, the greatest increase in frequency was observed in the thin/burn. However, at the quadrat scale (1 m<sup>2</sup>) there was no evidence of differential performance among treatments in cover or frequency. This could be because the species was relatively rare, occurring on an average of less than two quadrats per treatment unit. *Tragapogon dubious* can quickly invade disturbed sites (Merrill et al. 1980), and can remain in the community long after a disturbance is over, although at low densities (Gross and Werner 1982). However, few negative ecological or economic consequences have been documented for this species.

The final species to show differential performance among treatments was *Lactuca serriola*, which occurs throughout the U.S. and Canada and rapidly colonizes disturbed, open sites (Weaver and Downs 2003). Differences among treatments were only significant at the plot-level scale (1000 m<sup>2</sup>). This species responded by increasing

frequency most in the burn-only treatment, although it also increased in the thin/burn as well. There was no increase in frequency at any scale in the thin-only or control. While *L. serriola* is a common weed, there is little research showing detrimental environmental effects.

### Exotic impact

While this study presents evidence that restoration treatments, especially thinning combined with burning, may facilitate invasions by exotic species, there is no measure of the impact that these species may have on the system. Just increasing richness or cover of exotic species does not necessarily equate to a detrimental environmental impact, as not all exotic species are invasive (Mack et al. 2000; Williamson and Fitter 1996). While evidence suggests that the thin/burn was the most invaded treatment, the cover of all exotic species combined was less than two percent in this treatment, and less than one percent for the overall study. This may be contrasted with the cover of all native species combined (25.2 percent), which was more than an order of magnitude larger than exotic cover. This indicates that the overall impact of exotic species may be small.

Additionally, results from the regression analysis indicate that both native and exotic species are able to “invade” the same places. All treatments showed an average increase in native species richness after treatment that was greater than the exotic species richness increases. The control had an average of almost 10 additional native species for each new exotic, while the thin/burn had an additional three native species for each new exotic. Climatic data (Figure 8) shows that the spring months in 2003 were moister than 2000 or 2001 (NCDC 2004). This could partially explain why so many additional species, both native and exotic, were found after treatment that were not found prior to

treatment, as dry years may reduce understory plant richness relative to wet years (Fulé et al. 2002).

#### Future monitoring and analyses

This study documents changes in the exotic plant community two growing seasons after thinning and one season after burning. As such, this study presents only short-term results. It is unknown how the plant communities will change over time. In some instances exotics may be short-lived and quickly succeed to native species (Petryna et al. 2002; Gross and Werner 1982; D'Antonio and Meyerson 2002). However, problematic invaders are capable of causing severe environmental and economic harm (Vitousek 1996; Skinner et al. 2000), and may alter the long-term successional pathway of a site (D'Antonio and Meyerson 2002). Whether or not exotics maintain dominance in the plant community--they may contribute heavily to the seedbank--which could result in an even stronger response after future stand manipulations (Halpern et al. 1999; Alpert et al. 2000; Mitchell et al. 1998).

Propagules of the invader must be present for an invasion to take place (Hobbs and Huenneke 1992; Lonsdale 1999). This study made no attempt to measure propagule pressure. Roads have been found to increase invasion as they may provide corridors for exotic species to enter a community (Gelbard and Belnap 2003; Tyser and Worley 1992; Pauchard and Alaback 2004; Parendes and Jones 2000). However, there are also studies that show that the distance to roads has no effect on invasion by exotics (Harrison et al. 2002; Fornwalt et al. 2003). While the blocked design in this study ensures that no treatment unit is much further from a road than another, this factor could still explain

some additional variance in exotic species invasion, and this possible effect should be evaluated in the future.

Conditionality needs to be explored to find out what contexts prevent or facilitate invasion so they can be manipulated in management activities to favor native species. While treatments in this study represent the main alternatives available to land managers, there are many different ways of implementing them. The harvesting system used in this study was a cut-to-length system, which leaves slash on site. This increases fuel levels and may increase the severity of prescribed burning (Pollet and Omi 2002; Fahenstock 1968), potentially increasing invasion. Indeed the results of this study indicate that burning was more intense in the thin/burn, and that a more intense burn is associated with increased exotic richness. Other harvest techniques or slash management techniques may be more successful in reducing invasion.

The seasonality of the burn may also be important, as the effects on exotic species have been shown to differ depending on the time of burning (Tu et al. 2000). Fall burns would be more comparable to the historical fire regime (Wright and Agee 2004) and may be more beneficial for native species. However, at least one undesirable species at the Lubrecht study site, *Potentilla recta*, is favored more by fall burns than spring (Lesica and Martin 2003).

Petryna et al. (2002) found that fire increased exotic invasions in Argentinean grasslands. However, continued disturbance was necessary to maintain exotic species, because the majority of exotics were gone after three years. Fire also favored rare subordinate native species by enhancing their regeneration, so it was concluded that fire could be beneficial in these ecosystems despite facilitating exotic invasion (Petryna et al.

2002). It may also turn out that the benefits in restoring ponderosa pine forests outweigh the negative impacts of exotic invaders, especially if invasions are only temporary.

Paradoxically, doing nothing or applying treatments that are not effective in reducing fire hazard may lead to even more invasion after stand-replacing wildfire.

## **Conclusion**

The primary objective of this study was to evaluate exotic plant responses to restoration/fuel reduction treatments [control (no treatment), burn-only, thin-only, and thin/burn]. Undesirable status was applied to exotic species that are known to be especially harmful. Plant responses were estimated by frequency, richness, and cover, and sampled at multiple scales (1 m<sup>2</sup>, 100 m<sup>2</sup>, 1000 m<sup>2</sup>) to better detect treatment differences.

The pre-treatment data served as a baseline, with changes in vegetation from pre- to post-treatment being compared among treatments where possible. The control (no treatment) provided an estimate of year-to-year variability not due to treatment, which provided a basis to compare plant responses due to treatments. The control did show some change in exotic species richness, frequency, and cover from 2001 to 2003. These changes may be due to natural inter-annual variability in climate and other physical factors, as well as variability in field crew observations.

Exotic species responses the first full growing season after burning and the second full growing season after thinning indicate significant differences among treatments. The thin/burn consistently had the highest abundance for both overall undesirable and exotic species abundance, while the control generally had the least. Other restoration treatments were intermediate in exotic response.

Significant treatment differences were also evident in the response of individual species. Two undesirable species (*Cynoglossum officinale*, and *Potentilla recta*) responded most strongly in the thin-only. Indicator Species Analysis also indicated that

*Poa pratensis* responded to the thin-only more than the control. *Logfia arvensis* and *Lactuca seriola* responded strongly in the burn-only. No species that were significantly different among treatments in cover or frequency had their greatest abundance in the control. Additionally, no exotic species was a significant indicator of the control.

The majority of individual species that were significantly different among treatments had their highest cover or frequency in the thin/burn. Indicator Species Analysis revealed that prior to treatment no exotic species were indicators of the thin/burn when compared to the control. However, *Logfia arvensis*, *Poa pratensis*, *Verbascum thapsus*, *Cirsium vulgare*, *Taraxacum officinale*, and *Trifolium* species were all indicators of the thin/burn treatment. Additionally, *Bromus tectorum*, *Cirsium arvense*, *Carduus nutans*, *Polygynum aviculare*, and *Tragapogon dubious* all showed evidence of differential responses to treatment, each with their greatest abundance in the thin/burn.

While overall patterns in exotic species abundance did not differ among treatments the response of individual species did – depending on the scale. Cover was only estimated at multiple scales for the eight undesirable species. Therefore, patterns of cover for individual species did not vary much at different scales. However, frequency is dependent on the area sampled, and as such the large-plot (coarse-scale) measurements detected treatment differences for rare species, while the small-plot (fine-scale) measurements detected treatment differences for common species.

Environmental variables were examined to ascertain which attributes were associated with exotic invasion. The experimental block and four environmental variables were significant in explaining variation in the change of exotic richness from

pre-to post-treatment. Increases in exotic richness were positively correlated with decreased canopy completeness, increased height of bole charring, greater proportional removal of saplings, and increased native species richness. These variables combined explained 62 percent of the variation in change in exotic species richness.

Restoration treatments had strong effects on the understory exotic plant community in the short term. In most cases exotics increased in abundance from pre-to post-treatment, although the response was strongest in the thin/burn. Other factors, such as distance to roads and land management/usage history, may explain additional variability in invasion not quantified by this study. Continued monitoring of treated areas should provide insight as to the long-term dynamics of exotic species response to restoration treatments in ponderosa pine/Douglas-fir forests.

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## Appendices

**Appendix 1:** Overstory and sapling characteristics before and after treatment, and percent canopy cover estimates based on densitometer measurements and fish-eye photography.

Block	Treatment	pre-treatment			post-treatment			Live canopy cover (%) densitometer	Canopy cover (%) fish-eye
		Trees per ha	Basal Area m <sup>2</sup> /ha	Saplings per ha	Trees per ha	Basal Area m <sup>2</sup> /ha	Saplings per ha		
1	thin-only	325	19.26	974	116	9.62	826	40.00%	55.72%
1	burn-only	508	24.19	530	471	23.24	408	68.89%	71.42%
1	thin/burn	295	15.81	490	137	9.51	76	40.00%	52.21%
1	control	344	16.02	2790	337	15.87	2940	67.78%	69.18%
2	thin-only	451	20.92	1398	179	12.71	1058	42.22%	61.80%
2	burn-only	475	25.63	1666	351	21.18	1128	59.44%	72.99%
2	thin/burn	425	23.22	1270	142	9.48	318	36.67%	52.72%
2	control	424	24.10	1362	422	24.06	1726	67.22%	76.22%
3	thin-only	397	21.57	1644	153	10.60	1306	50.00%	56.06%
3	burn-only	343	18.49	4352	323	18.01	2394	67.78%	69.78%
3	thin/burn	346	23.58	1404	101	9.28	36	31.11%	45.19%
3	control	425	31.40	1820	415	31.14	2232	75.00%	80.01%

**Appendix 2:** Burn intensity measures and burn date for each of the six burned treatment units

Block	Treatment	Char cover (%)	Burn intensity	Burn effect	Crown scorch (%)	Bole char (m)	Scorch height (m)	Burn Date
1	burn-only	47.45	1.28	1.04	13.64	1.06	9.39	5/15/2002
1	thin/burn	67.22	1.58	1.39	29.36	2.91	9.04	5/1/2002
2	burn-only	65.76	1.98	1.78	30.51	1.59	7.42	6/25/2002
2	thin/burn	57.73	1.57	1.46	28.48	2.58	10.70	6/6/2002
3	burn-only	84.29	2.17	1.99	10.37	0.77	4.25	6/14/2002
3	thin/burn	88.08	2.28	2.20	43.92	4.31	13.21	5/31/2002

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