UNDERGROWTH VEGETATION RESPONSE TO FUEL REDUCTION TREATMENTS IN THE BLUE MOUNTAINS OF EASTERN OREGON

by
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Undergrowth Vegetation Response to Fuel Reduction Treatments

in the Blue-Mountains of Eastern Oregon

Director: Carl Fiedler

Undergrowth vegetation response to fuel reduction treatments was tested in fire-adapted ponderosa pine (*Pinus ponderosa*)/Douglas-fir (*Pseudotsuga menziesii*) forests. Treatments included: no treatment, prescribed burning, low thinning, and low thinning followed by prescribed burning. Direct effects of the fuel reduction treatments were observed on the undergrowth vegetation the first growing season after burning and three seasons after thinning. Burn-only treatments tended to reduce the diversity of the undergrowth and diminish the cover of grasses and shrubs while augmenting the frequency of fire adapted undergrowth species. Thin-only treatments had very little impact on species diversity but graminoids and shrubs tended to increase cover. Fire sensitive species were able to increase frequency in the Thin-only units. The Thin-and-Burn treatments elicited a response similar to that of the Burn-only treatments, though there was some indication that the fire may have been more intense, thereby magnifying the effect of burning.

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Introduction

Contemporary low elevation ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) forests exist in a weakened, fire prone condition, brought about by fire exclusion policies favored for the last 100 years (Covington *et al*. 1997; Smith and Arno 1999; DOI 2000). Thinning treatments and prescribed burning have been suggested to simulate or return historic disturbance regimes to ecosystems dependent on fire (Mutch *et. al.* 1993, Smith and Arno 1999; Covington *et al.* 1997). Fuel reduction is an important aspect of this process, and the Fire/Fire-surrogates (FFS) Project was funded by the Joint Fire Sciences Program, to assess the most effective methods of reducing fire hazard and restoring ecosystem structure and process in long-needled conifer forest. The intent of this study is to assess the effects of fire and fire-surrogate treatments on the undergrowth component of ponderosa pine/Douglas-fir forest communities.

In order to return historic structure and function to the forest, four treatments were considered: mechanical thinning, prescribed burn treatments, a combination of thinning and prescribed burns, and an untreated control. Some approximation of these treatments is being implemented at 11 sites across the country. This study is part of the Hungry Bob FFS Project located in the ponderosa pine and Douglas-fir forests of the Blue Mountains of northeastern Oregon.

One of the primary objectives of fire and fire-surrogate treatments is to reduce the potential for catastrophic fires. Increased resource availability (*e.g.*, water, light, nutrients) due to thinning and fire often has dramatic consequences for the diversity, distribution, cover, and species composition of the undergrowth. In the ponderosa pine forests of central Oregon, prescribed broadcast burning has been shown to increase species richness and diversity of the undergrowth while decreasing shrub cover (Busse *et* *al.* 2000). Prescribed burning increased undergrowth productivity while increasing the dominance of grasses over forbs in the ponderosa pine forests of Arizona (Harris and Covington 1983). Ayers *et al.* (1999) have reported that prescribed burning increases Scouler's willow (*Salix scouleriana*) while decreasing bitterbrush (*Purshia tridentata*), though thinning appears to have the opposite affect. Thinning alone has been shown to increase the cover of undergrowth in western Montana (Smith and Arno 1999), and dramatically increase the cover of grasses in eastern Washington (McConnell and Smith 1970). While undergrowth response to thinning and burning has been investigated in many systems, the opportunity to directly contrast undergrowth response to a variety of fuel reduction treatments in a replicated, completely randomized experiment in the ponderosa pine/Douglas-fir forests of northeastern Oregon is truly unique.

Specific objectives of this study include:

1) Comparing numeric indexes of species richness and evenness in the undergrowth vegetation among treatments
2) Identifying possible trends and short-term treatment effects on undergrowth vegetation response variables:
    a) Average cover (average cover of a species in each sample plot)
    b) Frequency (probability of species occurrence in a given sample plot)
3) Developing techniques for continued investigation into undergrowth response to fire and fire-surrogate treatments

Literature review

Fire has played a major role in many forest ecosystems (Franklin and Dyrness 1973; Hall 1977; Mutch *et. al.* 1993; Johnson *et. al*. 1998; Smith and Arno 1999). In much of the American West, fire has historically created and maintained relatively open stands of ponderosa pine (*Pinus ponderosa*) in which mean fire return intervals (the average length of time between reburns) can range from several to tens of years (Hall 1977; Bork 1984; Covington *et al.* 1997; Smith and Arno 1999). Primary effects of fire include: accelerated nutrient cycling, fire-stimulated germination of seeds, and increased heterogeneity of age classes and forest structure (Ahlgren 1960; Christensen and Muller 1975; Hall 1977; Harris and Covington 1983; Mutch *et. al.* 1993; Smith and Arno 1999). Due to a vigorous fire exclusion policy, this key disturbance factor has been largely removed from these ecosystems, resulting in insect infestations, disease outbreaks, and relatively frequent catastrophic wildfires. Returning historic disturbance regimes to ecosystems which evolved with fire could improve their vigor and overall health, while reducing the occurrence of catastrophic wildfires (Franklin and Dyrness 1973; Mutch *et. al.* 1993; Covington *et al.* 1997; Smith and Arno 1999; DOI 2000).

Overstory changes are perhaps the most obvious symptoms of the removal of disturbance from the ponderosa pine ecosystem, but concurrent with density, structure, and species composition shifts in the overstory, undergrowth vegetation has changed as well (Hall 1977; Mutch *et al.* 1993; Covington *et al*. 1997; Smith and Arno 1999). Prior to widespread fire suppression, the undergrowth of low-elevation ponderosa pine forests could have been typified as a bunchgrass savanna supporting a wealth of species and abundant forage (Hall 1977; Mutch *et al.* 1993; Covington *et al*. 1997; Smith and Arno

1999). Encroachment by, and in some cases a species shift in, coniferous regeneration has resulted in a less diverse and less vigorous undergrowth community (Hall 1977; Mutch *et al.* 1993; Covington *et al*. 1997; Smith and Arno 1999).

<u>Treatment options</u>

Potentially disastrous alterations to forest conditions due to fire exclusion have been widely documented (Franklin and Dyrness 1973; Hall 1977; Mutch *et. al.* 1993; Smith and Arno 1999; DOI 2000). In fact, 80 percent of Montana's 3.7 million hectares (ha) of fire-adapted forest land is rated high or moderate for crown fire hazard rating (Fiedler *et al.* 2001a). Though there may be a consensus that low-elevation ponderosa pine/Douglas-fir (*Pseudotsuga menziesii*) forests are unhealthy and that there is a need to reintroduce disturbance, opinions vary as to the most desirable management options (Covington *et al*. 1997; DOI 2000; Fiedler *et al*. 2001).

Forest fuels have been allowed to accumulate to extremely high levels with the result that the return of natural fire without intermediate treatments could result in stand replacement fires throughout many of our forests (Mutch *et al.* 1993; DOI 2000; Fiedler *et al*. 2001b). While stand-replacement fires could return these ecosystems to a baseline state from which a healthier condition might ensue in the long term, immediate consequences would generally be unacceptable. Potential loss of natural resources and private property, damage to unique ecosystems, and future fire suppression costs suggest that interventions to return short-interval disturbance to fire-adapted ecosystems could be in order (Mutch *et. al.* 1993; Smith and Arno 1999; DOI 2000).

Prescribed burning is one alternative by which elevated fuel levels may be reduced. By igniting fires at specific locations and under selected conditions, it is possible to return fire and associated ecosystem functions to the landscape in a more directed, less destructive fashion (Mutch *et al.* 1993; Smith *et al*. 1997; Busse *et al.* 2000). Successful prescribed burning in ponderosa pine/Douglas-fir forest types can thin the understory, recycle nutrients, and eliminate weak trees from the overstory, resulting in a more fire-resistant forest of large, healthy trees with a diverse and vigorous undergrowth (Harris and Covington 1987; Smith *et al*. 1997).

There are risks and complications associated with prescribed burning, however. Perhaps the greatest fear in prescribed burning is that the fire will escape and become a wild fire, with all of the negative implications discussed previously. Additional complications include smoke management and the difficulty of maintaining the fire at desired levels (Smith *et al*. 1997). Optimally, prescribed burns would take place at the time of year when natural ignitions are most abundant. However, duff and canopy fuels have accumulated to such a degree that prescribed burning can only take place in the fall or spring to guard against an escaped prescribed fire (Moore *et al.* 1999). Consumption of unnaturally high fuel loads in a time of year when fires do not naturally occur can lead to mortality of the older component of the stand due to girdling and fine root mortality (Swezy and Agee 1991). In addition, off-season burning can have ecological impacts on everything from conifer regeneration to undergrowth vegetation response (Ahlgren 1960; Ohmann and Grigal 1981; Enright and Lamont 1989).

An alternative to prescribed broadcast burning is the use of mechanical treatments to return a more fire resistant, historic structure to the forest. In addition to being less risky to implement, mechanical methods for manipulating stand structure are more flexible and may provide revenue for the landowner (Smith *et al.* 1997; Fiedler *et al*.

2001). Mechanical treatments can remove ladder fuels and reduce overstory density to deter the advance of a traveling crown fire, while invigorating the residual stand and inducing regeneration of vegetation dependent upon more open forest conditions (Hall 1977; Smith *et al.* 1997; Fiedler *et al*. 2001).

Thinning alone may not be sufficient to create a lasting, fire-resistant structure. Thinned stands allow shade-tolerant conifer regeneration to thrive, creating undesirable conditions which could be kept in check with more frequent, low-intensity fires (Gruell *et. al.* 1982; Smith and Arno 1999). Such shifts in coniferous species composition can affect undergrowth composition and overstory structure (Deal 2001). Deprived of fire, thinned stands can again become choked with regeneration of shade tolerant coniferous species. Undergrowth diversity, forest vigor, wildlife forage, and likelihood for catastrophic fire can potentially return to undesirable pre-treatment levels (Gruell *et. al.* 1982; Smith and Arno 1999).

A silvicultural treatment combining mechanical treatments and prescribed burning has potential to return both the structure and the function of fire-adapted ecosystems in a relatively safe, financially sound, and ecologically friendly manner. By first reducing the danger of a crown fire with mechanical means, burning is made less hazardous (Covington *et al*. 1997; Smith *et al.* 1997). Once fuels are sufficiently reduced through thinning and burning treatments, future treatments to preserve a more historic forest structure and function could take the form of prescribed burns during the more ecologically appropriate fire season (Moore, *et al.* 1999). Frequent, low-intensity fires can maintain an open canopy and promote greater micro site heterogeneity in which relatively rare species can thrive, increasing species richness, as opposed to only a few common species achieving dominance (Spies and Franklin 1989; Covington *et al.* 1997; Smith and Arno 1999).

Undergrowth response

Undergrowth vegetation is of particular interest because of its sensitivity and relatively rapid response to variations in site resources (Pfister and Arno 1980; Nieppola 1992). Immediately after disturbance, undergrowth vegetation has been shown to be a sink that captures nutrients which could otherwise be lost from the system (Harris and Covington 1983). In addition, diversity of the undergrowth and wildlife has become an increasingly important resource to scientists and laypersons alike (Thomas 1979; Smith and Arno 1999). Though the concept is somewhat controversial, if a diverse system is more stable, as suggested by MacArthur (1955), then a forest system with greater biodiversity (in the undergrowth) should be more resilient to stochastic deleterious events.

Canopy closure due to the removal of disturbance leads to suppression of undergrowth vegetation, and thus decreased diversity and forage. A particularly vivid example of this phenomenon occurs in old clearcuts of southeast Alaska. Without subsequent management actions to control density, coniferous regeneration results in virtual extirpation of undergrowth species due to competition for light after canopy closure (Alaback and Herman 1988). In contrast, periodic disturbance in the form of partial cuttings can lead to a structure which more closely resembles that of old-growth forests, and supports greater undergrowth diversity (Deal 2001).

A similar response to canopy closure has been noted, though to a lesser extent, in once-open stands of ponderosa pine (Hall 1977; Mutch *et. al.* 1993; Smith and Arno

1999). In addition to competition for light, however, water and other belowground resources are particularly limiting in ponderosa pine forests (Riegel, Miller, and Krueger 1992). In the absence of frequent natural underburns, reduced forage production was reported in the Blue Mountains of eastern Oregon, primarily due to decreased pinegrass (*Calamagrostis rubescens*) and elk sedge (*Carex geyeri*) abundance (Hall 1977). Fuel reduction treatments that free up resources for the undergrowth could reverse this trend (Mutch *et. al.* 1993; Covington *et al.* 1997; Smith and Arno 1999).

Studies in thinned-only plots show a resultant flush in vegetation with increased response in grasses with increased tree spacing (McConnell and Smith 1970; Conway 1981; Bedunah *et. al.* 1988). This response to reduced canopy density makes sense in light of historic photos showing relatively open stands dominated by large trees with undergrowth dominated by bunchgrasses (Smith and Arno 1999). Observations in the Blue Mountains suggest that the majority of trees were historically open grown, and grasses were a more prevalent feature of the forest community (Hall 1977). Some similarities of thinned stands to the historic vegetative community do not necessarily imply that thinning treatments are sufficient to return historic structure and function to the forest community.

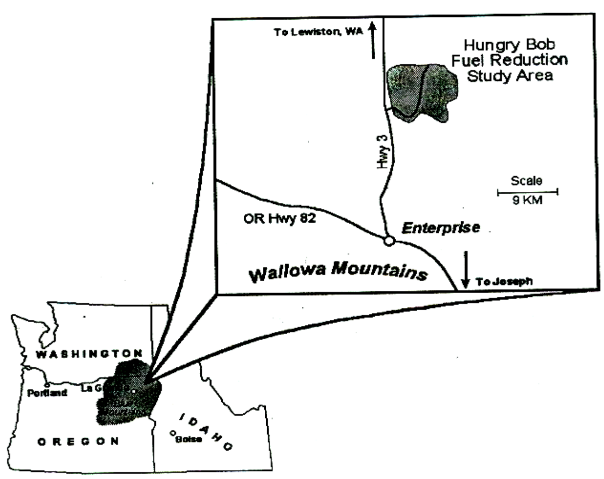
Magnitude and duration of response to burning treatments is often more pronounced and longer lasting than to thin-only treatments (Dyrness 1973; Abrams and Dickman 1982). Additionally, burning favors fire-adapted species such as Scouler's willow while reducing the cover of fire sensitive species such as bitterbrush and Idaho fescue (*Festuca idahoensis*) (Harris and Covington 1983; Ayers *et al.* 1999; Smith and Arno 1999; Busse *et al*. 2000). Fire-adapted species often rely on the creation of microsites for seedling success, stimulation of sprouting, or on heat-or smoke-induced germination to ensure seedling growth under conditions of reduced competition and increased resource availability (Ahlgren 1960; Christensen and Muller 1975; Harris and Covington 1983; Smith and Arno 1999). As a result of fire favoring particular adaptations and creating heterogeneous micro sites, species richness is often higher in burned areas and the species composition and dominance is often different between burned areas and those which were mechanically thinned (Dyrness 1973; Abrams and Dickman 1982; Ayers *et al.* 1999).

Understanding the relationships between fuel reduction treatments and undergrowth vegetation may allow forest managers to predict changes in abundance and distribution of undergrowth plants (McKenzie *et al.* 2000). As long as there is a need for fuel reduction treatments (Mutch *et al.* 1993; Smith and Arno 1999; DOI 2000; Fiedler *et al.* 2001), and a desire to manage multiple facets of the forest community (Thomas 1979; Smith and Arno 1999), there is a need to understand how the undergrowth vegetation will respond. By conducting controlled experiments under operational conditions, extrapolation to management situations will be more reliable and ultimately more useful. This opportunity to experimentally test potential fuel reduction treatments in fire-adapted ponderosa pine/Douglas-fir forests is unique and potentially quite useful.

Study site

This study detailed the response of the undergrowth vegetation on the Hungry Bob Fire and Fire Surrogate (FFS) project, one of 11 research sites in a national network. The research area was located in the Blue Mountains of northeastern Oregon on the Wallowa Valley Ranger District between the Davis and Crow Creek drainages, 45 kilometers north of Enterprise, Oregon (Figure 1). Average yearly temperature is 6° Celsius (43° Fahrenheit) with an average of 146 frost-free days (Weatherbase 2000). The thirty year average annual precipitation is 49.9 centimeters (cm), the majority of which falls between September and June (National Climate Data Center (NCDC) 2002).

Figure 1**:** Location of Hungry Bob FFS research site, 45 km north of Enterprise, Oregon (Youngblood 2000).



Since its initiation, the Hungry Bob FFS project has become increasingly complex as additional components have been added. Initially, the project was simply a study of fuel reduction treatments and their effects on the overstory, underground processes, and economics. With the creation of the national FFS project, additional facets of the forest community, such as wildlife, insects, and a complete undergrowth census have become a part of the larger project (Youngblood 2000). Now into its sixth year, numerous research-related activities have been accomplished on the Hungry Bob FFS project (Table 1).

Table 1: Timeline of treatments and measurements.

Season and Year	Activity
Winter 1996	"Alternative fuel reduction methods in Blue Mountain dry forests"
Summer 1996	Site selection
Fall 1997	Timber sale design
Winter 1998	Unit layout and grid establishment
Spring-Summer 1998	Pre-treatment measurements
Summer-Fall 1998	Treatment: mechanical thinning
Fall 1999	Joint fire science program funding for national FFS network approved
Summer 2000	Thinning post-treatment measurements
Fall 2000	Treatment: prescribed fire
Summer 2001	Re-measurement of all treatment units

Study Design

Stands were selected and treatments allocated using a completely randomized design (Table 2). The stands were randomly selected from a large pool of ponderosa pine (*Pinus ponderosa*)/Douglas-fir (*Pseudotsuga menziesii*) stands that were relatively homogenous in regard to slope, aspect, elevation, basal area, plant association, and Stand Density Index (SDI; Reineke 1933). Treatments were randomly assigned at the stand level, with treatment units located within representative portions of the larger treatment areas. A grid of sampling points was then established within each treatment unit. These points were 50 meters (m) apart and at least 50 m from stand edges.

Table 2: Listing of unit numbers and corresponding treatments. Units within brackets were considered as one treatment.

Treatment	Unit Numbers
Control	(2, 4, 5), 15, 18, 23
Thin-only	6A, 7, 9, 22
Thin-and-Burn	6B, 8A, 10A, (11, 12)
Burn-only	8B, 10B, 21, 24

Treatments

Both thinning and prescribed burning were designed to reduce basal area from about 27.5 m^2/hectare (ha) to about 16 m^2/ha. Thinning was prescribed to reserve dominant and codominant crown classes. Natural clumping was enhanced. All live trees greater than 32 cm diameter at breast height (DBH; 1.37 m) were left standing and any trees growing within 9 m of dominant trees were removed in order to accentuate structural characteristics of the stand. Harvested trees were limbed, and the slash was trampled by the harvester and left in place (Youngblood 2000).

Prescribed burning prescriptions were designed to allow survival of set percentages of pre-treatment basal area. Survival targets for trees between 20 and 51cm DBH were, 70-80 percent ponderosa pine, 60-80 percent Douglas-fir, and up to 30 percent grand fir (*Abies grandis*). For trees larger than 51 cm DBH, target basal area survival percentages were 80 percent for ponderosa pine, 70 percent for Douglas-fir and 50 percent for grand fir. Fuel bed mass was targeted for reduction to less than 21, 800 kilograms/ha of material less than 8 cm in diameter (Youngblood 2000).

Field Methods

Physiographic characteristics of each plot were assessed during the pre-treatment measurements. At each point, aspect was obtained to the nearest 1° azimuth using a compass, and slope was obtained to the nearest 1° inclination using a clinometer. Topographic position was classified it into one of the following categories: ridge top, convex slope, even slope, concave slope, swale, bottom of a slope, or on a flat. Elevation of each site was obtained from USGS contour maps to the nearest 15 m. Soils were typed and mapped.

Circular, 200 m^2 reconnaissance plots (radius 8.0 m) were used for measurement of the pretreatment undergrowth cover as well as assessment of the pretreatment overstory. After further consideration the 200 m^2 reconnaissance plot was deemed an inadequate sample, and so circular 400 m^2 reconnaissance plots (radius 11.3 m) were used for the remainder of the study. These plots were centered on every grid point, approximately 25 per treatment unit.

Percent cover of all vascular plants was estimated ocularly to the nearest 1 percent up to 10 percent and to the nearest 10 percent for all values greater than 10 percent. A plant did not need to be rooted in the plot in order to contribute cover. Plant identification was conducted in the field using Johnson (1998), but with nomenclature and more specific identification according to Hitchcock and Cronquist (1973).

In order to accurately characterize the overstory canopy cover, observations were taken with a moosehorn densitometer 2 m from the plot center in each of the four cardinal directions, as well as one observation at the plot center. Each observation in which live foliage was viewed was tallied. Overstory canopy cover for the treatment unit was then derived as a percent using equation 1:

[1] $$Canopy\,cover = \frac{X}{N}$$

Where X is the number of observations in which foliage was viewed and N is the number of possible observations within the unit.

All trees within each sample plot were identified by species and assessed as live or dead. Breast height diameter (1.37 m) was measured to the nearest 0.1 cm using a diameter tape. Height was calculated to the nearest 0.1 m using either a clinometer or a telescoping height pole. Cover of all advance tree regeneration (seedlings ≤ 1.37 m^2 in height) was estimated ocularly to the nearest 1 percent. In order to assess fire intensity, vertical bark char along the bole was measured to the nearest 0.1 m. Percent of area with mineral soil exposed and consumption of woody fuels were also recorded. Crown scorching for each tree was ocularly estimated to the nearest 5 percent of total crown.

Summary Methods

Response variables

Several response variables were identified in order to address the questions laid out in the objectives. Biodiversity was investigated using three response variables: species richness, Shannon's index of diversity (Shannon and Weaver 1949) and Pielou's index of evenness (Pielou 1975). Two vegetative response variables were employed in order to identify possible trends and short-term treatment effects on undergrowth vegetation growth and distribution: average cover and frequency.

The indices of diversity were simple enough to be used efficiently and yet have proven to be effective measures of diversity and evenness. Pielou's evenness index is not completely independent of species richness (Smith and Wilson 1996). In other words, with evenness held constant, the index value of J′ increases as richness increases. This was not a problem because the number of species in each unit was above 25. One alternative measure of evenness is E_{var} (Camargo 1993), which, according to Smith and Wilson (1996) is more equally sensitive to minor and abundant species, and is independent of species richness. J′ is more commonly used (Smith and Wilson 1996; Chiarucci *et al.* 1999), however, so this study utilized J′ as well.

Two data sets were employed in this analysis. The first set consisted only of the undergrowth data collected in 2001. Objectives had changed by the sixth year of the

Hungry Bob project; instead of simply identifying key species, all species present were identified. This set consisted of 191 reliably identified undergrowth species--species which had an unknown number or common name for reference. Differences in diversity among treatment units were tested using this data set. The second data set spanned all three years of sampling but consisted of only 29 species, mostly grasses, which were reliably identified in 1998. Vegetative changes throughout the years of the study were investigated with this reduced dataset.

Diversity

Due to the intensive nature of the sampling in 2001, it was possible to use numeric indices to represent the undergrowth community and to compare those index values among treatments. Species richness was the number of individual species found in a unit. Shannon's diversity index, H′ (Shannon and Weaver 1949), was used to add a cover component to this straight-forward measure, giving a representation of undergrowth abundance:

[2] $$H' = -\sum_{i=1}^{i} p_i \, ln(p_i)$$

Where p_i is the proportion of the community belonging to the i'th species and ln is the natural logarithm. In this case, percent cover was used as a representation of p_i.

In order to more effectively isolate the distribution of aboveground cover between species, Pielou's evenness index, J′ (Pielou 1975) was used:

[3] $$J' = H'(ln(S))^{-1}$$

Where H´ is Shannon's index as in equation 2, ln is the natural logarithm and S is the number of species in the sampled unit.

Vegetative characteristics

Two response variables were calculated in order to quantify the vegetative response to fuel reduction treatments. The first vegetative response variable was average cover for each species (Appendix 6). This value was the average percent cover for a species for each sampling plot in a unit for a specific year. In addition, frequency was calculated to describe the frequency of occurrence of each species on the landscape. Frequency was calculated by unit as the number of sample plots in which a species was found, divided by the total number of sample plots in the unit (Appendix 7).

Instead of investigating how cover and frequency of the subset of 29 species changed, the goal of the data analysis was to evaluate how the vegetative community responded to treatments. For the 1998-2001 vegetation data, dimension reduction was desired due to the number of response variables. To this end, principal components analysis (PCA) was used to suggest combinations of undergrowth species which would best explain the variability in the data set. This step was not necessary with the diversity indices calculated for the 2001 data set because there were only two variables.

A PCA was run for both vegetative response variables (average cover and frequency) utilizing a covariance matrix for all 29 species over all three years of measurement. For each of the response variables, as many as three principal components were identified in an attempt to explain at least 70 percent of the variability in the data with the initial eigenvalues (Table 3).

Table 3: Variance explained by principal components for the response variables average cover and frequency.

Response variable	Principal Component	Initial Eigenvalues		
		Total	% of Variance	Cumulative %
Average cover	1	117.35	47.70	47.70
	2	57.97	23.57	71.27
Frequency	1	0.31	27.66	27.66
	2	0.27	24.44	52.10
	3	0.15	13.77	65.87

Once the required principal components were identified, they were related to the data set to increase interpretability. This was accomplished by analyzing the component coefficient matrix (Appendix 1) to deduce which species contributed most strongly to explaining the variability in the data. Component coefficients between -0.2 and 0.2 did not contribute enough to the principal components to be considered in the analysis (Steele 2002).

Independent parameters

Before reliable predictive linear equations could be derived, independent parameters had to be identified, and if necessary, modified or transformed for use in the analysis. Physical site variables were mostly useful as recorded. Aspect, however, had to be modified to be useful for modeling purposes in order to convert from circular values (0 to 360) to linear values (1 to -1). Many parameters were measured in an attempt to characterize the coniferous component of the vegetation. After evaluating many possible independent parameters, it became evident that only a few were actually significant. Parameters which were not been significant in any of the models evaluated were dropped, and only eight independent parameters were used for the remainder of the analysis (Table 4).

Table 4: Mean parameter values actually used to calculate response variable adjusted means. All seedlings are conifers less than 1.37 m in height. Saplings have a DBH between 0.01 cm and 10 cm. Overstory trees are conifers with a DBH greater than 10 cm.

Parameters	**Mean**	**Standard error**
Elevation (m)	1271.903	23.844
Effective aspect	-0.012	0.017
Slope (percent)	12.876	1.309
Soil: Percent Bocker	0.250	0.112
Soil: Percent Fivebit	0.375	0.125
Soil: Percent Larabee	0.063	0.063
Soil: Percent Melhorn	0.188	0.101
Soil: Percent Olot	0.125	0.085
Pre-treatment percent SDI	36.998	2.461
Pretreatment overstory cover (percent)	55.166	2.634
Pretreatment sapling crown ratio	40.465	2.323
Pretreatment seedling crown ratio	46.895	4.297

Physical parameters

Assumptions of independence, linearity, and constant variance among the physical parameters were investigated using Spearman's *rho* and a scatterplot matrix. Strong correlation was observed between treatment and elevation, aspect, slope, and soil, while no strong non-linear relationships were evident. This implied some bias between the treatments, and justified the use of multivariate statistical methods in the analysis to account for these discrepancies. Some of the parameters, such as elevation and aspect, were correlated as well. These correlations indicated the need for either an interaction variable or the elimination of certain parameters in the multivariate analysis. This was accomplished by eliminating parameters which were not statistically significant ($p<0.05$) from all general linear models. Additionally, some heteroscadasticity was evident, suggesting that scrutiny of the residual plots of any ensuing mathematical models would be necessary.

Assumptions of normality in regards to the distribution of the numeric physical parameters across treatments were investigated using box plots. Two of the physical parameters, topography and soils, were categorical variables so non-parametric figures were necessary to determine normality.

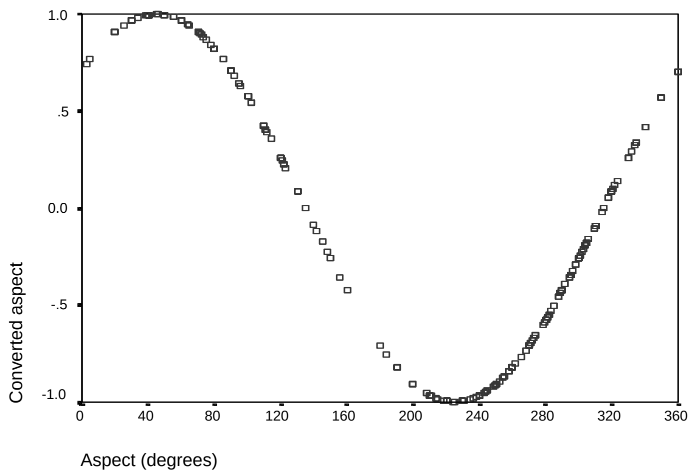
Elevation was the most normally distributed of the physical characteristics. Elevation means were reasonably similar among treatments, as the inter-quartile ranges all overlapped. The biggest difference was between the Thin-only and the Thin-and-Burn treatments. Mean elevation for the Thin-and-Burn treatment was exceptionally low, but skewed quite strongly to the higher elevations.

To effectively work with the data for aspect, the data had to be converted in a manner similar to that utilized by Stage (1976):

[4] Converted aspect = cosine ((aspect in degrees) - 45^o)

The correlation between aspect and plant growth was maximized by subtracting 45^o from the original values, which were taken in degrees. The cosine of the resulting values was then taken in order to create a data set which was linear (1…-1) instead of circular (0…360). The resulting values showed the greatest negative effect of the sun on the southwest aspect (225^o) with a value of –1, and the least negative effect on the northeast aspect (45^o) with a value of 1 (Figure 2).

Figure 2: Recorded plot aspect data plotted against



Converting aspect to cosine and shifting the center of its distribution linearized the aspect data and made it more normal. However, the true effect of aspect is not captured without accounting for slope. As the slope increases, aspect has a greater influence on the vegetation of the site (Stage 1976). Converted aspect values were multiplied by the tangent of slope angle to obtain a value better representing the true effect of aspect:

[5] Effective aspect = (Converted aspect) * tangent (angle of slope)

A box plot of the effect of aspect revealed that these values were nearly normally distributed, with a mean of zero. The Thin-and-Burn treatment exhibited a slightly negative effect of aspect, meaning that steeper slopes and more southwestern aspects were associated with this treatment. The Control and Burn-only treatments demonstrated a normal distribution of converted aspect, unlike the Thin-only and the Thin-and-Burn treatments, which were somewhat skewed but with no outliers.

Slope means were very consistent across the treatments. Skew was reasonably uniform across the treatments, and did not require remedial action. As an important factor in determining the influence of aspect, slope was included as an interaction variable with aspect in addition to being included for other influences it may have had on vegetation development.

Due to the categorical nature of the topographic data, it had to be investigated in a non-parametric fashion. A bar chart was fashioned which allowed a relatively efficient visualization of how topographic features were distributed throughout the study. The vast majority of the sites were even. Other topographic features were not well enough represented to be very useful in making inferences about treatment effect.

The soils data were also categorical and had to be investigated in a non-parametric fashion. Using the 'crosstabs' function in SPSS 9.0, the soil series found in every plot was summarized for each treatment. All of the treatments were very heterogeneous in regard to soil series. Fivebit was the most prevalent soil series, although the percentage it comprised varied from only 22 percent in the Control treatment, to 51 percent in the Thin-and-Burn treatment. The Thin-and-Burn treatment had the most consistent soil series (51 percent Fivebit) followed by the Burn-only treatment (44.3 percent Fivebit). Soil series varied the most in the Control treatment with nearly equal proportions of Fivebit (22 percent) and Melhorn (23.9 percent).

Physical characteristics were averaged by unit and treatment in order to give an idea of the variability within the research site (Table 5). The non-parametric data could not be averaged; instead, the mode (most frequently occurring value) was used. Elevation averages varied considerably among units, but were similar among treatments.

This was not the case with average converted aspect. However, after the effect of slope was accounted for, effective aspect was relatively constant among treatments, suggesting that treatment units were fairly flat. Slope was also somewhat variable across the units but relatively constant when averaged by treatment. Topographic feature was constant with an even slope. Soils data were perhaps the most variable.

Table 5: Plot physical characteristics by unit and treatment.

Treatment	**Unit**	**Average elevation (m)**	**Average converted aspect**	**Average effective aspect**	**Average Slope (percent)**	**Topographic mode**	**Soil Mode**
Control	15	1113	0.4052	0.0727	17	Even	Melhorn
Control	18	1333	-0.2996	-0.0674	25	Even	Melhorn
Control	23	1412	0.3562	0.0328	9	Even	Olot
Control	245	1286	-0.3007	-0.0186	11	Even	Fivebit
Burn	10B	1192	0.376	0.0363	7	Even	Bocker
Burn	21	1374	-0.9335	-0.1718	18	Even	Fivebit
Burn	24	1260	0.5645	0.0626	9	Even	Bocker
Burn	8B	1169	0.6606	0.0532	8	Even	Olot
Thin	22	1380	-0.3335	-0.0311	8	Even	Larabee
Thin	6A	1361	-0.4818	-0.0499	11	Even	Fivebit
Thin	7	1305	-0.3429	-0.0686	20	Even	Fivebit
Thin	9	1235	0.9553	0.1179	12	Even	Melhorn
Thin and burn	10A	1186	-0.2677	-0.0222	13	Even	Bocker
Thin and burn	1112	1183	-0.5630	-0.0545	9	Even	Bocker
Thin and burn	6B	1388	-0.5808	-0.0983	15	Even	Fivebit
Thin and burn	8A	1174	-0.2257	-0.0143	8	Even	Fivebit
Control	---	1297	0.0690	0.0079	15	Even	Melhorn
Burn	---	1258	0.0833	-0.0179	11	Even	Fivebit
Thin	---	1324	-0.0830	-0.01147	13	Even	Larabee
Thin and burn	---	1239	-0.4239	-0.0503	12	Even	Fivebit

Conifer characterization

Undergrowth response to treatments was more accurately ascertained by evaluating several conifer characteristics and testing for significance. First, tree data were separated into three size classes: seedlings (trees less than 1.37 m tall), saplings (trees greater than 1.37 m tall, up to 10 cm DBH), and overstory trees (DBH greater than 10 cm). Tree densities in terms of basal area (BA) and trees per ha were calculated by first deriving the value for each plot, then averaging over all of the plots in the unit or treatment to get an overall value and associated standard error (Appendix 2).

[6] $$BA(m^2) = DBH(cm)^2 \times (7.854 \times 10^5)$$

Stand Density Index (SDI) was chosen as a potential explanatory variable based on its proven utility in predicting undergrowth production in previous studies (McKenzie *et. al.* 2000; Moore and Deiter 1992). Deal (2001) found that undergrowth plant community structure was strongly tied to stem density, which was strongly tied to the species composition of the overstory. While SDI is not a direct measure of density, SDI incorporates overstory species and stem relative density for a measure more representative of site utilization than stems per hectare or basal area.

SDI was calculated based on maximum stockings and species-specific SDI equations (Table 7) using the summation technique (Long 1996). Percent SDI was then derived by comparing the actual SDI of each species to the maximum possible for that species (Fiedler 2002). The use of SDI as a percent allowed for a more accurate representation of conifer resource utilization by accounting for differing rates of resource consumption among conifer species. Once a percent SDI was calculated for each species, these values were summed to derive the total percent SDI for each plot, and then averaged by unit and treatment (Appendix 2).

[7] $$SDI_{Spp} = \sum_{N=1}^{i} (DBH_i / 25.4cm)^{b_{Spp}}$$

Where DBH_i is the diameter at breast height (cm) of the i'th tree of the species and b_{Spp} is a species specific constant (Table 6).

$$[8] \qquad SDI\%_{Spp_i} = (SDI_{Spp} / SDI_{SppMax}) * 100$$

Where SDI_{SppMax} is a predetermined species-specific maximum Stand Density Index.

$$[9] \qquad SDI\%_{Total} = \sum_{i=1}^{i} SDI\%_{Spp_i}$$

Table 6: Values used for calculation of percent SDI.

Species	b_{Spp}	SDI_{SppMax} (Trees per hectare)	Source
Pinus ponderosa	1.77	901.5	De Mars and Barrett 1987
Pinus contorta	1.74	684.2	File data
Larix occidentalis	1.73	1012.7	Cochran 1985
Pseudotsuga menziesii	1.51	938.6	Seidel and Cochran 1981
Abies grandis	1.73	1383.2	Cochran 1983
Picea engelmannii	1.73	1158.4	Estimated

Table derived from Cochran *et. al.* (1994) Table 1.

Trees per hectare, BA, percent SDI, and overstory cover were calculated by size class and plot (Appendices 2-4). Other parameters that were calculated included: average height by size class, average percent canopy by size class, and percent cover of the overstory as determined using a moosehorn densitometer. These values were also calculated on a plot-by-plot basis and then averaged over unit and treatment (Appendix 5).

Analytical methods

Treatment effects on the undergrowth vegetation were investigated by performing analysis of variance on the adjusted means of the relevant undergrowth characteristics (response variables). Characteristics that were evaluated included: species richness, Shannon's index of diversity (H′), Pielou's index of evenness (J′), average cover, and frequency. If response variables were not normal in distribution, a natural logarithmic transformation was used. Adjusted means were obtained via regression equations formed using stepwise forward and backward multiple linear regressions (Ott 1993), a technique favored by Brosofske *et al.* (2001) for analyzing changes in undergrowth richness, and

McKenzie *et al.* (2000) for investigating overstory influences on undergrowth vegetation. These adjusted means were then tested statistically to determine if changes observed were significant, or simply the result of the many uncontrollable variables inherent in any natural experiment.

In order to adjust for differences among units, elevation, effective aspect, slope, soil series (Bocker, Fivebit, Larabee, Melhorn, and Olot), and pretreatment coniferous data were tested for significance with response variables in a general linear model. Because treatment and soil series were categorical, dummy variables were used to represent them. The other four parameters were continuous random variables.

Parameters were not included if the significance (p-value) of their coefficient (β) was greater than 0.05. Treatment, year, and a treatment X year interaction variable were dealt with as fixed factors in the model and never eliminated, regardless of significance. An extra sums of squares F-test, utilizing the method of least squares, was conducted to determine if the dummy variables for soil were significant; they were subsequently retained or eliminated as a group (Ott 1993). When the most parsimonious model had been derived, response variables were described only by the set of parameters which explained a significant portion of their variability; highly correlated or insignificant parameters were not included in the final models.

Once the best fitting model had been obtained, adjusted mean values were calculated for each treatment using parameter values averaged over the entire study (Table 7). Another extra-sums of squares F-test utilizing the method of least squares (Ott 1993) was run in order to determine if treatment had a significant influence on the response variable. Response variable adjusted means were then tested for significant differences using a student's t-test based on the probability of observing the calculated difference between treatments if the response variable was actually the same between treatment units. Additionally, 95 percent confidence intervals were calculated for the adjusted mean values of the response variables. In order to avoid pseudoreplication, all analyses were conducted using unit averages, giving a sample size of four per treatment or sixteen total.

Table 7: Mean parameter values used to calculate response variable adjusted means. Seedlings are all conifers shorter than 1.37 m in height. Saplings have a DBH between 0.01 cm and 10 cm. Overstory trees are conifers with a DBH greater than 10 cm.

Parameters	**Mean**	**Standard error**
Elevation (m)	1271.903	23.844
Effective aspect	-0.012	0.017
Slope	12.876	1.309
Soil: Bocker	0.250	0.112
Soil: Fivebit	0.375	0.125
Soil: Larabee	0.063	0.063
Soil: Melhorn	0.188	0.101
Soil: Olot	0.125	0.085
Pre-treatment percent SDI	36.998	2.461
Pretreatment overstory cover	55.166	2.634
Pretreatment overstory TPH	307.771	29.228
Pretreatment overstory BA	17.357	1.103
Pretreatment overstory height	16.234	0.466
Pretreatment crown ratio	45.314	1.025
Pretreatment sapling TPH	155.504	33.292
Pretreatment sapling BA	0.306	0.065
Pretreatment sapling height	3.966	0.277
Pretreatment sapling crown ratio	40.465	2.323
Pretreatment seedling TPH	220.349	55.862
Pretreatment seedling height	0.626	0.098
Pretreatment seedling crown ratio	46.895	4.297

Results

Each of the fuel reduction treatments--Control, Burn-only, Thin-only, and Thin-and-Burn--elicited a unique response from the response variables. Biodiversity of the undergrowth vegetation was investigated first using species richness, which is the number of species found in each unit. Shannon's index of diversity (H′), a measure of the number of species and the overall area covered, was utilized to more robustly portray the contribution of the undergrowth to the forest community. Pielou's index of evenness was used to obtain a measure of how the above ground cover was distributed among the species. Principal components analysis (PCA) was used to streamline the analysis of changes in overall cover and distribution of individual undergrowth species. By investigating closely only those species and groups of species identified by the PCA as most representative, the analysis was made more informative and efficient.

PCA suggested that the variability in average cover was best described by changes in the graminoids. In particular, pinegrass (*Calamagrostis rubescens*) and elk sedge (*Carex geyeri*) provided the most explanation of variability, 47.7 percent. Due to the apparent usefulness of investigating graminoids as a assemblage, and the pervasiveness in the literature of analyzing undergrowth vegetation by lifeform (Harris and Covington 1983; Bedunah *et al*. 1988; Busse *et al.* 2000; Deal 2001), vegetative cover was analyzed first by lifeform, i.e., graminoids, forbs, and shrubs. Principal components analysis suggested that additional insight could be gained when the combined values of pinegrass and Idaho fescue (*Festuca idahoensis*) were contrasted against values for elk sedge; 23.6 percent of the variability in cover was explained by this interaction.

Frequency, which is the percent of sample plots in which a given species was identified, varied in a fashion that was not as easily explained. The first principal component derived to explain variability in the frequency data only explained 27.7 percent. This component was almost exclusively comprised of western yarrow (*Achillea millefolium*) and elk sedge frequency. The second component explained 24.4 percent of the variation, almost as much as the first component. It appeared to be a contrast between elk sedge and the combined constancies of western yarrow, Idaho fescue, and prairie Junegrass (*Koelaria macrantha*). The third component only explained 13.5 percent of the variability in the frequency data, and appeared to be a contrast between the frequency response of arrowleaf balsamroot (*Balsamorhiza sagittata*) and western needlegrass (*Stipa occidentalis*). The three principal components did not point to any logical grouping of species, although several response tendencies were identified by investigating these components.

Biodiversity

Fire and fire surrogate treatments had the least effect on numeric diversity measures of any of the response variables investigated. In 2001, species richness, the number of species per unit, was found to be lowest in units that received fuel reduction treatments. Shannon's index of diversity (H′) provided a measure of the number of undergrowth species and the percent cover occupied by those species. Trends identified using Shannon's index of diversity suggested that fuel reduction treatments reduced the cover occupied by the undergrowth in addition to decreasing the number of undergrowth species. In particular, a decrease in H′ was implied in response to treatments involving prescribed burning. Pielou's index of evenness (J′), which represents the proportion of aboveground cover belonging to each species of undergrowth vegetation, changed very little in response to treatment. Because changes in evenness were not statistically significant, possible general trends were all that could be identified.

Species richness

Species richness, which is the number of species per unit, was lower in the treated units than in the Control units in 2001 (three growing seasons after thinning and the first growing season after burning). By first accounting for differences in pretreatment overstory cover among treatment units with a general linear model, 0.570 (R^2) of the variability in species richness was explained (Table 8), and treatment effects were more clearly isolated. As a group, treatments were significant in the model (p = 0.032).

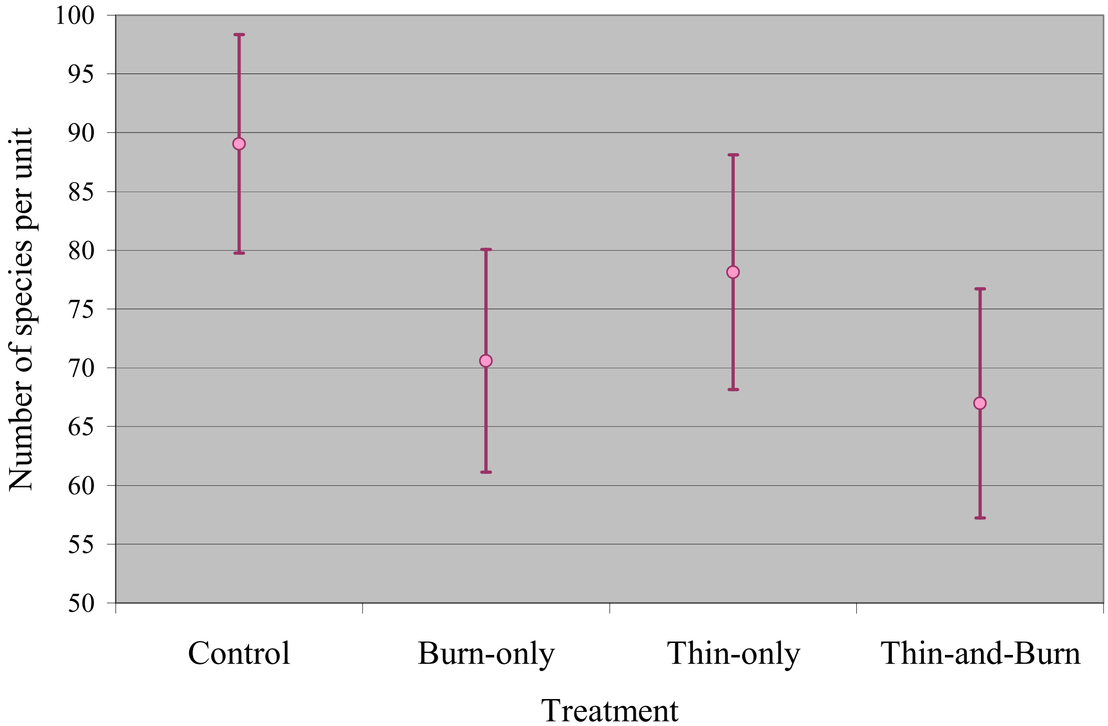
Table 8: Linear model used to adjust treatment means for species richness. Parameter coefficients (β), significance and 95 percent confidence intervals are also presented. Only parameters significant at the 95 percent level were included.

Parameter	β	Significance	95 percent Confidence Interval	
			Lower Bound	**Upper Bound**
Intercept	106.430	0.000	77.330	135.531
Control	22.077	0.006	7.656	36.499
Burn-only	3.601	0.536	-8.815	16.017
Thin-only	11.151	0.138	-4.200	26.501
Thin-and-Burn	0.000[a]	.	.	.
Pretreatment overstory cover	-0.715	0.020	-1.296	-0.135

a This parameter is set to zero because it is redundant.

Adjusted mean species richness was significantly reduced in those treatments which received burning, relative to the Control units (Figure 3). Thin-only treatments appeared to reduce the number of undergrowth species relative to the control, though there were more species in the Thin-only than in the Burn-only and Thin-and-Burn treatment units. The lowest species richness was observed in the Thin-and-Burn treatments.

Figure 3: Adjusted mean species richness by treatment for 2001 with associated upper and lower bounds for 95 percent confidence intervals.



Thin-and-Burn treatment units had 22 fewer species than the Control units ($p = 0.006$) in 2001 (Table 9). Burn-only treatment units also had significantly reduced species richness relative to the Control as determined using a 2-tailed t-test ($p = 0.015$). Species richness in the Thin-only units was slightly lower than in the Control units, but this difference was not statistically significant ($p = 0.081$). Differences in species richness among treated units were not statistically significant; the greatest difference was that the Thin-only units had 11 more species than the Thin-and-Burn units ($p = 0.138$).

Table 9: Differences in species richness among treatments (treatment I – treatment J) with associated tests for significance and 95 percent confidence intervals for the difference.

(I) Treatment	(J) Treatment	Mean Difference (I-J)	Significance	95 percent Confidence Interval for Difference	
				Lower Bound	Upper Bound
Control	Burn-only	18.5*	.015	4.395	32.557
Control	Thin-only	10.9	.081	-1.584	23.437
Control	Thin-and-Burn	22.1*	.006	7.656	36.499
Burn-only	Thin-only	-7.5	.290	-22.502	7.404
Burn-only	Thin-and-Burn	3.6	.536	-8.815	16.017
Thin-only	Thin-and-Burn	11.1	.138	-4.200	26.501

Based on estimated marginal means
* The mean difference is significant at the .05 level.
a Adjustment for multiple comparisons: Least Significant Difference (equivalent to no adjustments).

Shannon's index of diversity (H´)

All of the treatments had lower diversity than the Control in 2001. The linear model (Table 8) which best fit H′ explained 0.740 (R^2) of the observed variation, and was used to adjust H´ treatment means to account for differences among the 16 treatment units. Slope was the only independent variable which was significant in the model (Table 10), meaning that by accounting for slope with the model, treatment effect was more clearly isolated. Though the Burn-only and the Thin-only treatments were not significant individually, fuel reduction treatments influenced H´ as a group ($p = 0.049$).

Table 10: Linear model used to adjust treatment means for H´. Parameter coefficients (β), significance and 95 percent confidence intervals are also presented. Only parameters significant at the 95 percent level were included.

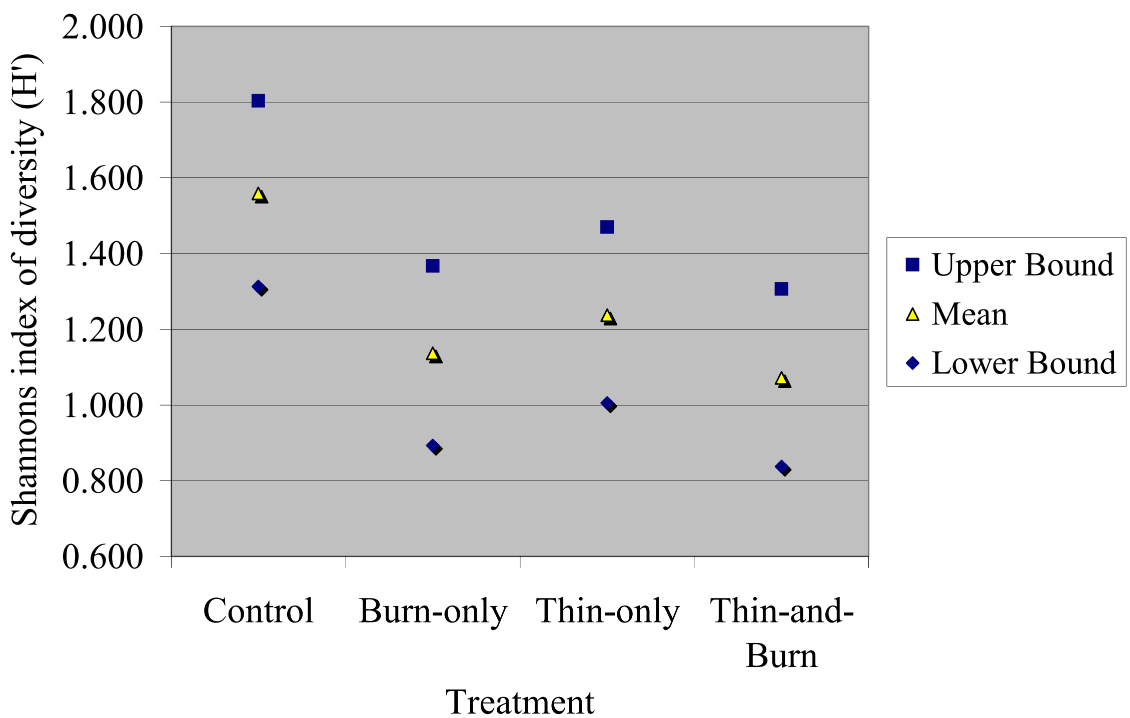
Parameter	**β**	**Significance**	**95 percent Confidence Interval**	
			Lower Bound	**Upper Bound**
Intercept	0.648	0.003	0.279	1.018
Control	0.487	0.010	0.139	0.834
Burn-only	0.058	0.704	-0.270	0.387
Thin-only	0.166	0.294	-0.165	0.496
Thin-and-Burn	0.000[a]	.	.	.
Slope	0.033	0.014	0.008	0.058

a This parameter is set to zero because it is redundant.

After adjusting H´ treatment means for differences among treatment units, H´ values varied little among the three treatments. The 95 percent confidence intervals for H′ overlapped substantially across treatments, suggesting that the results differed little

(Figure 4). Diversity in the Thin-and-Burn treatment appeared to be lower than any other treatment and 31 percent lower than the Control.

Figure 4: Adjusted mean H′ by treatment for 2001 with associated upper and lower bounds for 95 percent confidence intervals.



Closer analysis of the differences among treatments, which were tested for significance with a 2-tailed t-test, revealed that the Burn-only ($p = 0.022$) and the Thin-and-Burn units ($p = 0.010$) were statistically different from the Control (Table 11). The Thin-only treatment did not change the values for H´ in a statistically significant fashion, though there was a downward trend. The greatest decrease in diversity was associated with the Thin-and-Burn treatment, where H´ of 1.072 was much lower than the comparable index of 1.558 for the Control (Figure 4).

Table 11: Differences in H′ among treatments (treatment I – treatment J) with associated tests for significance and 95 percent confidence intervals for the difference.

(I) Treatment	(J) Treatment	Mean Difference (I-J)	Significance	95 percent Confidence Interval for Difference	
				Lower Bound	Upper Bound
Control	Burn-only	0.428*	0.022	0.075	0.781
Control	Thin-only	0.321	0.060	-0.015	0.657
Control	Thin-and-Burn	0.487*	0.010	0.139	0.834
Burn-only	Thin-only	-0.107	0.494	-0.440	0.226
Burn-only	Thin-and-Burn	0.058	0.704	-0.270	0.387
Thin-only	Thin-and-Burn	0.166	0.294	-0.165	0.496

Based on estimated marginal means
* The mean difference is significant at the .05 level.
a Adjustment for multiple comparisons: Least Significant Difference (equivalent to no adjustments).

Pielou's evenness index (J′)

None of the fuel reduction treatments influenced the distribution of aboveground undergrowth cover in a statistically significant fashion. Variability in J′ values was mostly explained by the linear model 0.793 (R^2), which was used to adjust for differences among treatment units (Table 12). Pretreatment overstory cover and effective aspect were the variables which significantly influenced values of J′. Effective aspect was the most influential variable with a parameter coefficient (β) of –0.097. Treatment variables were not significant as a group (p = 0.472), providing insufficient evidence to support the hypothesis that fuel reduction treatments influenced J′ values.

Table 12: Linear model used to adjust treatment means of J′. Parameter coefficients (β), significance and 95 percent confidence intervals are also presented. Only parameters significant at the 95 percent level were included.

Parameter	**β**	**Significance**	**95 percent Confidence Interval**	
			Lower Bound	**Upper Bound**
Intercept	0.004	0.825	-0.031	0.039
Control	0.009	0.287	-0.009	0.028
Burn-only	-0.003	0.693	-0.017	0.012
Thin-only	0.003	0.724	-0.016	0.022
Thin-and-Burn	0.000[a]	.	.	.
Pretreatment overstory cover	0.001	0.030	0.000	0.002
Effective aspect	-0.097	0.031	-0.183	-0.011

a This parameter is set to zero because it is redundant.

The adjusted means for J′ values, which account for variability among treatment units, suggest that burning decreased evenness, though only slightly, relative to the Control (Figure 5). A two-tailed t-test of the differences observed among treatments revealed that J′ values were not statistically different between any two treatments (Table 13). There was a trend implying that evenness decreased in units that were burned. In other words, prescribed burning appeared to have increased the competitive advantage of one or a few undergrowth species, whether or not in the presence of thinning. With that slight advantage, these species may have gained some dominance of the undergrowth cover.

Figure 5: Adjusted mean J′ by treatment for 2001, with whiskers for 95 percent confidence intervals.

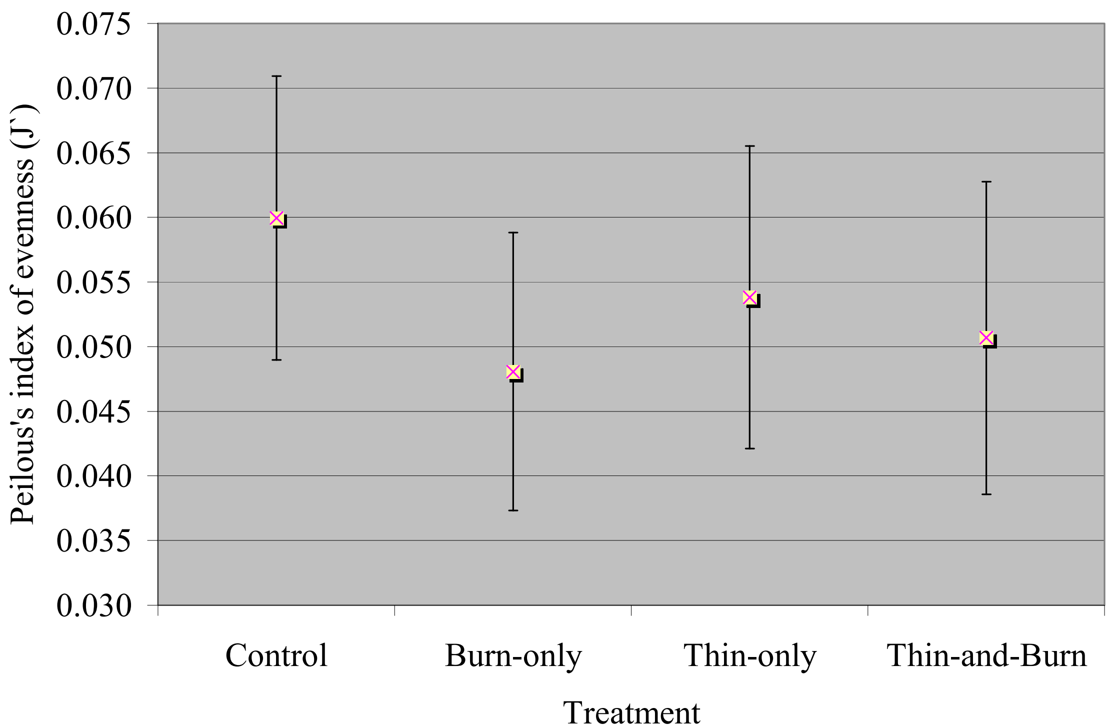


Table 13: Differences in J′ among treatments (treatment I – treatment J) with associated tests for significance and 95 percent confidence intervals for the difference.

(I) Treatment	(J) Treatment	Mean Difference (I-J)	Significance[a]	95 percent Confidence Interval for Difference	
				Lower Bound	Upper Bound
Control	Burn-only	0.012	0.143	-0.005	0.028
Control	Thin-only	0.006	0.353	-0.008	0.020
Control	Thin-and-Burn	0.009	0.287	-0.009	0.028
Burn-only	Thin-only	-0.006	0.482	-0.023	0.012
Burn-only	Thin-and-Burn	-0.003	0.693	-0.017	0.012
Thin-only	Thin-and-Burn	0.003	0.724	-0.016	0.022

Based on estimated marginal means

* The mean difference is significant at the .05 level.

a Adjustment for multiple comparisons: Least Significant Difference (equivalent to no adjustments).

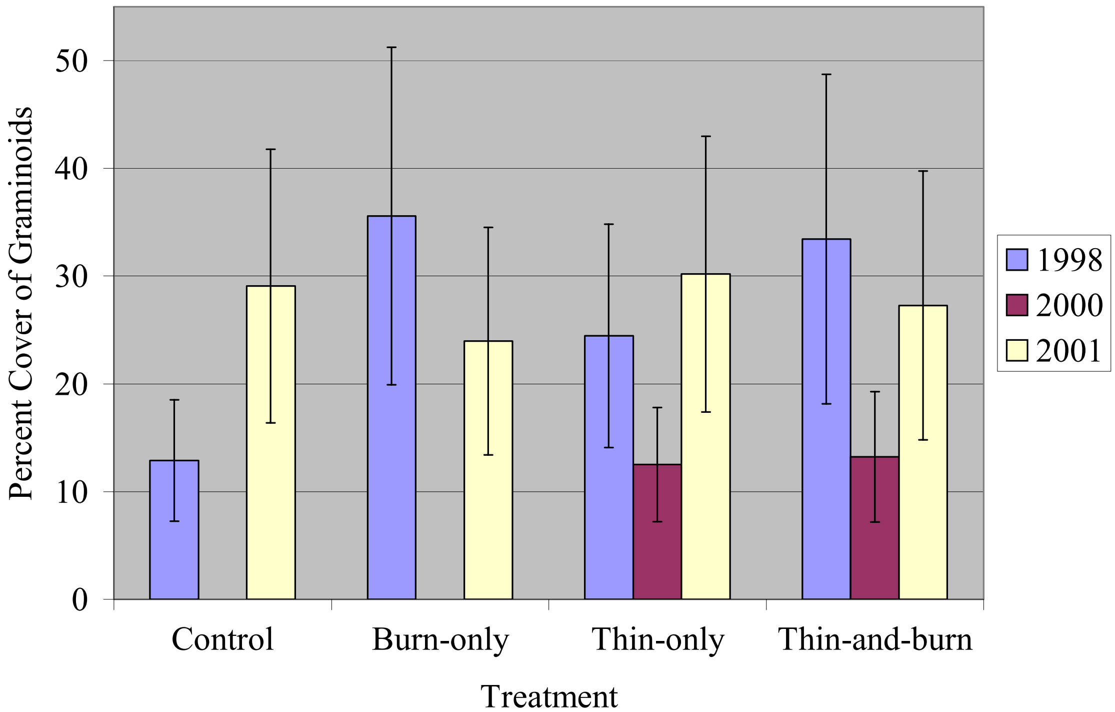
Cover

Cover of undergrowth vegetation was sensitive to fire and fire surrogate treatments. Response was quite different among lifeforms; graminoids, forbs, and shrubs were influenced differently by prescribed burning, low thinning, and low thinning followed by prescribed burning. A closer look at the effects of treatments on inter-specific dynamics of graminoid species revealed several species which appeared to respond most dramatically to fuel reduction treatments.

Graminoids

Fuel reduction treatments did not significantly affect the adjusted mean cover of graminoids. There was no significant difference in graminoid cover between 1998 and 2001 among the treated units (Figure 6). The passage of time, on the other hand, doubled graminoid cover in the Control units. Absence of this trend in the treated units suggested that perhaps treatments actually reduced cover. The Control units had the lowest adjusted mean cover in 1998, but in 2001, all of the treated units had approximately the same cover as the Control, Burn-only slightly less and Thin-only units slightly more. There was a decreasing trend in graminoid cover between 1998 and 2001 in response to burning, although this trend was not significant.

Figure 6: Adjusted mean graminoid cover by treatment and measurement year. Measurements were not taken in the Control and Burn-only units in 2000. Whiskers represent 95 percent confidence intervals.

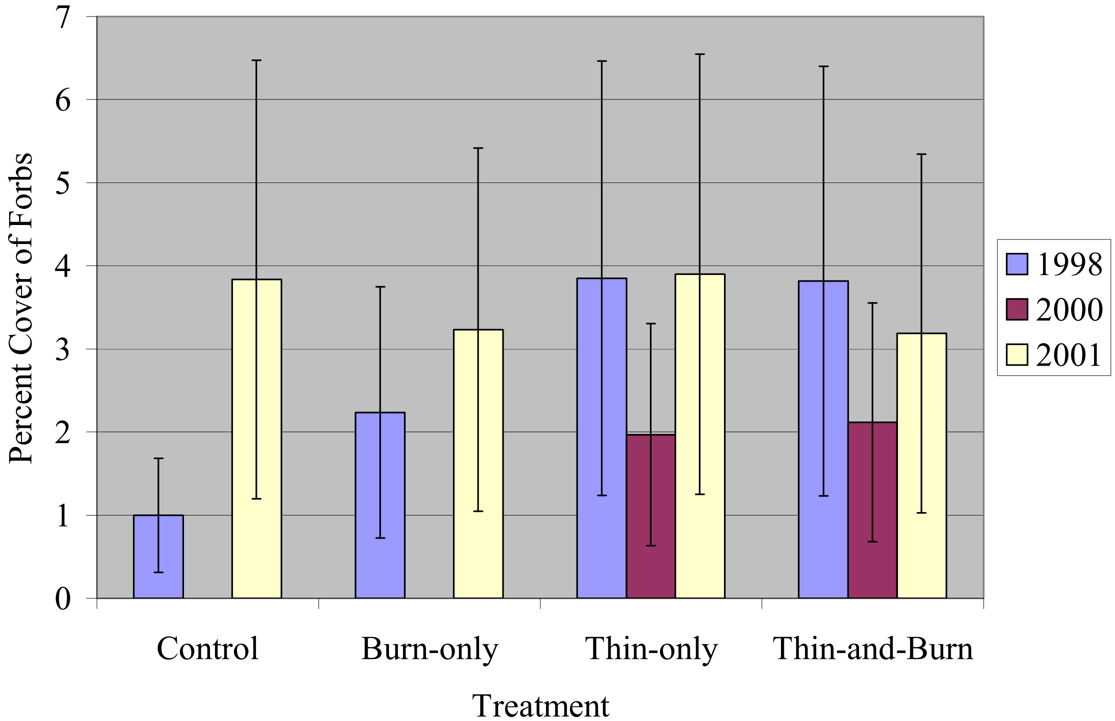


A dramatic increase in graminoid cover was observed between 2000 and 2001 in the units which were thinned in 1998. Low thinning reduced the cover of graminoids by about 50 percent two years post-treatment. Between 2000 and 2001, graminoid cover increased up to pretreatment levels. This recovery was observed even in thinned units which were subsequently burned.

<u>Forbs</u>

Forb cover increased between 1998 and 2001 without treatment, as demonstrated by the more than doubling in cover in the untreated units (Figure 7). The data suggest that in Burn-only units, forb cover increased by nearly half between 1998 and 2001. Two years post-treatment (2000), thinning treatments reduced overall forb cover by about 45 percent. One year later, in 2001, forb cover had responded in the Thin-and-Burn treatment units with an increase in cover of about half. The Thin-only treatment increased forb cover as well in 2001, though only by about 35 percent, to a level almost identical to pretreatment values.

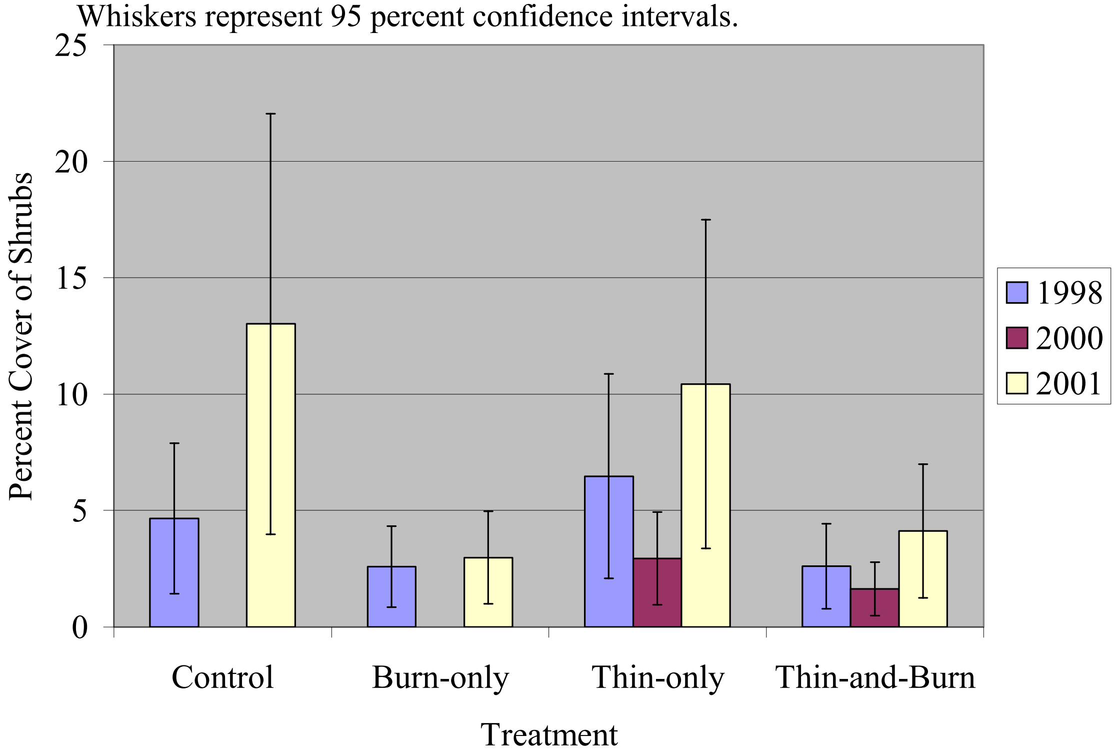
Figure 7: Adjusted mean forb cover by treatment and measurement year. Measurements were not taken in the Control and Burn-only units in 2000. Whiskers represent 95 percent confidence intervals.



Shrubs

As with total forb cover, overall shrub cover more than doubled in the Control units from 1998 to 2001 (Figure 8). Before treatment, the Thin-only units had the highest shrub cover at about 6 percent, while the Burn-only and the Thin-and-Burn treatment had the least with only 2.5 percent. Treatments that included burning exhibited only about 20 percent of the shrub cover of Control treatments in 2001. There was a clear trend toward increased cover in the Thin-only treatments, from about 6 percent in 1998 to 11 percent in 2001. A trend similar to that observed in the graminoids was evident in the shrub cover; thinning reduced shrub cover by half in the Thin-only units in 2000, only to increase in cover the next year to nearly twice the pretreatment levels. Prescribed burning appeared to dampen this response to thinning; shrub cover values for the Thin-and-Burn units in 2001 were only slightly higher than pretreatment levels.

Figure 8: Adjusted mean shrub cover by treatment and measurement year. Measurements were not taken in the Control and Burn-only units in 2000. Whiskers represent 95 percent confidence intervals.



Inter-specific interactions

A few undergrowth species, such as elk sedge (*Carex geyeri*) and pinegrass (*Calamagrostis rubescens)*, responded favorably to all fire surrogate treatments. Other species, such a prairie Junegrass (*Koelaria macrantha*) and Idaho fescue (*Festuca idahoensis*), were more responsive to fire and, while thinning induced a response, it was not as great as that observed when fire was introduced to the landscape (Appendices 6-7). Results of the principal components analysis (PCA) suggested that a relationship could exist among the covers of elk sedge, pinegrass, and Idaho fescue.

A variable was constructed based on the combined cover of pinegrass and Idaho fescue, minus the cover of elk sedge. In the second year after thinning (2000), elk sedge cover had increased relative to pinegrass and Idaho fescue by twofold in the Thin-only units and by fourfold in the Thin-and-Burn treatment units. An additional year of response in the Thin-only units resulted in values for this variable being 15 percent higher in 2001 than pretreatment levels. Treatments that included prescribed burning decreased values for this variable from 1998 to 2001 by 60 percent in the Burn-only units and 33 percent in the Thin-and-Burn units.

Closer investigation of the adjusted mean cover of these three graminoids helped explain some of the inter-specific dynamics which were observed in response to treatments (Table 14). The Burn-only treatment resulted in an 84 percent reduction in Idaho fescue cover between 1998 and 2001. Conversely, in the Burn-only units, elk sedge cover increased by twofold from 1998 to 2001. Thin-only treatments resulted in pinegrass cover that was 50 percent greater in 2001 than before the treatment. Treatments which involved burning actually reduced pinegrass cover from 1998 to 2001--by 15 percent in Burn-Only and 30 percent in the Thin-and-Burn units.

Table 14: Adjusted mean cover values with associated 95 percent confidence intervals (CI) for three graminoid species. Measurements were not taken in the Control and Burn-only units in 2000.

Species	Year	1998				2000		2001			
	Treatment	Control	Burn-only	Thin-only	Thin-and-Burn	Thin-only	Thin-and-Burn	Control	Burn-only	Thin-only	Thin-and-Burn
Elk sedge	Cover	2.88	2.12	0.96	1.31	0.69	1.93	15.08	5.15	2.64	1.79
	95 percent CI +/-	2.35	1.94	1.25	1.42	1.08	1.80	9.76	3.83	2.32	1.71
Pinegrass	Cover	3.09	12.15	12.46	25.71	6.63	8.22	8.17	10.38	17.81	18.20
	95 percent CI +/-	2.41	7.95	7.77	16.62	4.41	5.74	5.42	6.88	10.86	11.95
Idaho fescue	Cover	4.32	6.43	3.65	3.21	2.49	0.75	3.47	1.04	4.05	2.03
	95 percent CI +/-	3.82	5.16	3.36	3.12	2.52	1.30	3.21	1.42	3.65	2.25

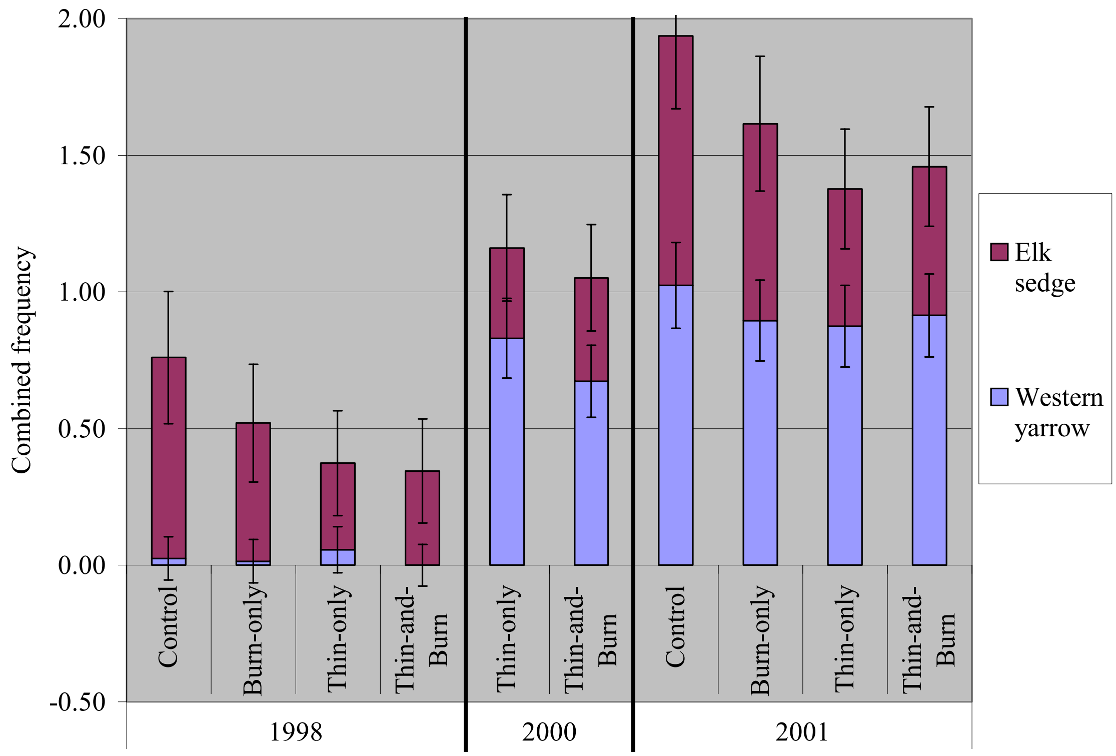
Frequency

Species distribution across the study units was affected by the fire and fire surrogate treatments in complex ways. This intricacy in frequency response was compounded by variations in frequency due to differences among years. Some species, such as western yarrow (*Achillea millefolium*) and elk sedge, tended to increase in frequency irrespective of treatment; others, such as prairie Junegrass and Idaho fescue, were quite sensitive to fire. While most species' frequency did not remain stable throughout the course of the study, the magnitude of change differed dramatically, as illustrated by the relationship between arrowleaf balsamroot (*Balsamorhiza sagittata*) and western needlegrass (*Stipa occidentalis*).

Yearly change

Some of the variability in the frequency data was explained by changes in western yarrow and elk sedge. The raw data suggest that these two species increased similarly from 1998 to 2001, regardless of treatment (Appendices 6-7). A variable created by combining the frequency of western yarrow and elk sedge, as suggested by the PCA, very clearly illustrated that trend; the combined constancies of these two species more than doubled over the course of the study in all treatments (Figure 9). The increase in frequency of these two species was observed in 2000, two years after implementing the thinning treatments, but was particularly evident in the Thin-only units in which the combined frequency increased by threefold. Between 2000 and 2001, the western yarrow and elk sedge frequencies in the Thin-and-Burn units responded vigorously to Burn-only treatments, increasing by about 3.6 times the pretreatment levels.

Figure 9: Adjusted mean frequency of western yarrow and elk sedge, with a maximum possible value of 2. Measurements were not taken in the Control and Burn-only units in 2000. Whiskers represent one standard error



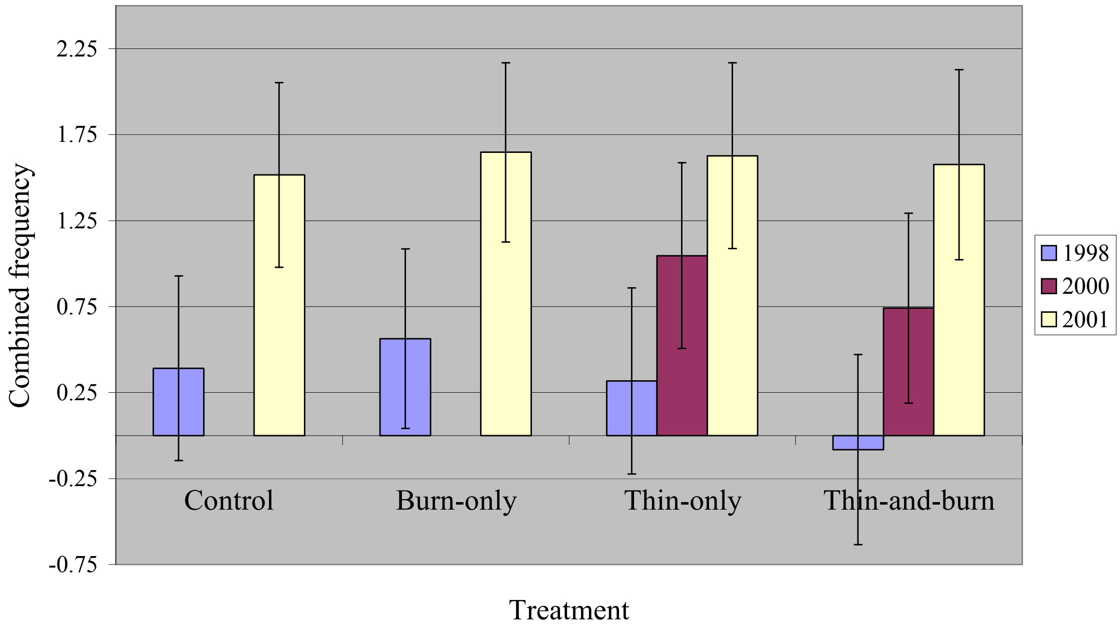
Charting the values for each species on the graph allowed for an easy comparison of how western yarrow and elk sedge frequency actually changed in response to treatments (Figure 9). Western yarrow frequency increased dramatically, irrespective of treatment. In 1998, western yarrow was found in very few plots, whereas in subsequent years, western yarrow was found in nearly every plot. By the year 2001, western yarrow was present in over 90 percent of the sample plots for all treatments.

After adjusting for site variation and conifer influences (Figure 9), the increases in elk sedge, suggested by the raw data (Appendices 6-7), were quite pronounced. A 23 percent increase in frequency was suggested between 1998 and 2001 in the Control units. Increases between 1998 and 2001 were fairly consistent across the treated units, with the greatest increase observed in the Thin-and-Burn units (58 percent) and the least in the Burn-only units (41 percent).

Treatment differences

The second most important source of variation in the frequency data, as identified by the PCA, was the combined constancies of western yarrow, Idaho fescue, and prairie Junegrass contrasted against the frequency of elk sedge. The data suggest that the values for this combined variable tended to become more positive, regardless of treatment (Figure 10), implying that elk sedge was less dominant in 2001 than in 1998 relative to western yarrow, Idaho fescue, and prairie Junegrass. A clear trend was observed in those units which received thinning treatments. In 2000, elk sedge dominance had been decreased by more than fourfold. This trend continued into 2001, where elk sedge dominance was reduced by another 2X from 2000 levels, regardless of prescribed burning.

Figure 10: Contrast between the adjusted mean frequency of elk sedge and the combined values for western yarrow, Idaho fescue, and prairie Junegrass. Smaller values are indicative of increased elk sedge dominance. Measurements were not taken in the Control and Burn-only units in 2000. Whiskers represent 95 percent confidence intervals.



Closer analysis of the adjusted mean frequency of the individual species (Table 15) suggested that elk sedge frequency was not decreasing. Instead, other species were becoming more prevalent in each treatment, resulting in the observed trend of decreasing elk sedge dominance. In the Control units, western yarrow frequency increased 50X and prairie Junegrass more than doubled. Prairie Junegrass also increased in the Burn-only units (80 percent) and in the Thin-and-Burn units (90X). Only western yarrow (14X) and arrowleaf balsamroot (3X) increased in response to thinning in 2000. An additional year of response allowed some of the other species to increase frequency over 2000 levels, most notably Idaho fescue by 1.5X and prairie Junegrass by 8.5X. The Thin-and-Burn treatment reduced the frequency of Idaho fescue by a moderate amount (14 percent) in 2001, but increased the frequency of elk sedge by 1.4X and prairie Junegrass by 7X over

2000 levels. Somewhat surprisingly, Idaho fescue, a fire-sensitive grass species (Gruell *et al.* 1982; Smith *et al.* 1999; Busse *et al.* 2000), decreased in frequency by only 1 percent in the Burn-only and 4 percent in the Thin-and-Burn units between 1998 and 2001.

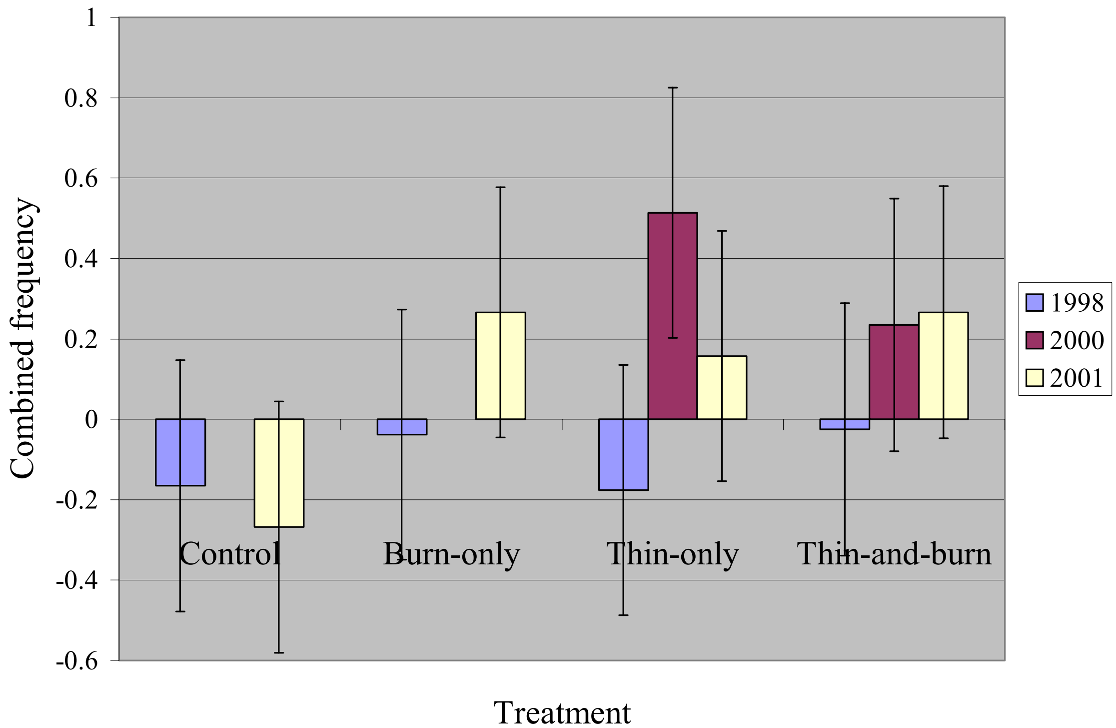
Table 15: Adjusted mean frequency with associated 95 percent confidence intervals (CI) for species identified through PCA as representative of overall changes. Measurements were not taken in the Control and Burn-only units in 2000.

Species	Year	1998				2000		2001			
	Treatment	Control	Burn-only	Thin-only	Thin-and-Burn	Thin-only	Thin-and-Burn	Control	Burn-only	Thin-only	Thin-and-Burn
Western yarrow	Frequency	0.02	0.01	0.06	-0.03	0.83	0.67	1.02	0.90	0.87	0.91
	95% CI +/-	0.08	0.08	0.08	0.08	0.15	0.13	0.16	0.15	0.15	0.15
Elk sedge	Frequency	0.74	0.51	0.32	0.34	0.33	0.38	0.91	0.72	0.50	0.54
	95% CI +/-	0.24	0.22	0.19	0.19	0.19	0.19	0.27	0.25	0.22	0.22
Idaho fescue	Frequency	0.53	0.73	0.44	0.62	0.42	0.58	0.57	0.72	0.66	0.53
	95% CI +/-	0.26	0.30	0.25	0.28	0.25	0.27	0.27	0.29	0.29	0.27
Prairie Junegrass	Frequency	0.20	0.43	0.05	0.01	0.06	0.13	0.47	0.81	0.51	0.91
	95% CI +/-	0.15	0.17	0.13	0.12	0.13	0.14	0.18	0.22	0.18	0.23
Arrowleaf balsamroot	Frequency	0.01	0.11	0.00	0.13	0.31	0.37	0.16	0.41	0.21	0.42
	95% CI +/-	0.15	0.16	0.13	0.17	0.20	0.20	0.17	0.21	0.18	0.21
Western needlegrass	Frequency	0.11	0.14	0.20	0.12	0.05	0.12	0.38	0.12	0.31	0.14
	95% CI +/-	0.14	0.15	0.15	0.15	0.13	0.15	0.18	0.15	0.16	0.15

Magnitude of response

A third approach was employed to help explain a significant amount of variability in the frequency data. Results of the PCA suggested that the relationship between the frequency of arrowleaf balsamroot and western needlegrass should be investigated further (Figure 11). All treatments increased the dominance of arrowleaf balsamroot over western needlegrass by at least threefold. Thinning had an exceptional influence on the cover of these two species in 2000, while the Thin-and-Burn treatment continued this trend. Treatment effects moderated somewhat by 2001, although this decline was less noticeable for treatments that included burning.

Figure 11: Contrast between the adjusted mean frequency of western needlegrass and arrowleaf balsamroot. Smaller or negative values are indicative of increased western needlegrass. Measurements were not taken in the Control or Burn-only units in 2000. Whiskers represent 95 percent confidence intervals



Investigation of the individual adjusted mean frequency for each species revealed that treatments mostly increased the frequency of both species, but with a much greater response from arrowleaf balsamroot (Table 15). In 2000, thinning had increased arrowleaf balsamroot from a frequency of 0.0 to 0.3 in the Thin-only units and nearly tripled its frequency in the Thin-and-Burn units. In 2001, arrowleaf balsamroot frequency decreased slightly in the Thin-only treatment, increased fourfold in the Burn-only, and increased 13 percent in the Thin-and-Burn, relative to 2000 levels, but still remained three times higher than prior to treatment. Western needlegrass had essentially no response in the Burn-only and the Thin-and-Burn treatments. This species increased to 38 percent frequency in the Control in 2001 and from 20 percent to a frequency of 31 percent in the Thin-only treatment. While both species tended to increase their presence in response to treatments, arrowleaf balsamroot frequency increased even more dramatically than did the frequency of western needlegrass.

Discussion

Results of this study differ somewhat from other reports in the literature on undergrowth vegetation response to silvicultural treatments. In particular, the analysis relating to numeric indexes of species diversity produced unexpected results. Dominant paradigms as to cover and frequency response to treatments held up better than those for diversity, though there were still some unique or unexpected outcomes. While not consistent with many numerous other investigations of undergrowth response to thinning and fire, results of this study are corroborated by some examples in the literature. Furthermore, these results provide an opportunity to critically evaluate the factors affecting response to disturbance and the methods used for investigating them.

Due to the relatively low intensity of the prescribed burning in this study, rhizomatous undergrowth species would be expected to respond quite vigorously, perhaps to the exclusion of invasive species (Stickney 1986; Grant and Loneragan 2001). The uneven pattern and intensity of burning would be expected to increase the number of potential niches and elevate the species richness of the overall community. In addition to increased richness (Busse *et al.* 2000), cover and frequency of undergrowth species, particularly graminoids (Harris and Covington 1983), would also be expected to increase as a response to burning treatments.

Units receiving low thinning would be expected to respond similarly, after a year or so of lag time for the species to respond to treatments. Diversity of the undergrowth could be expected to increase (Ahlgren 1960; Conway 1981). Cover and frequency of the undergrowth could have increased as well, particularly the graminoids (McConnell and Smith 1970). Rhizomatous species such as Scouler's willow (*Salix scouleriana*), pinegrass (*Calamagrostis rubescens*), and elk sedge (*Carex geyeri)*, could have capitalized on the newly available resources.

Undergrowth response to low thinning would be expected to be further amplified by subsequent prescribed burning. Indeed, changes in diversity and cover of the undergrowth were expected to be greatest in the Thin-and-Burn treatment units (Dyrness 1973; Abrams and Dickman 1982; Ayers *et al.* 1999). By burning two years after thinning, undergrowth vegetation released by thinning had an opportunity to build up rhizomes, seedbanks, and energy reserves before the burning occurred. This could have resulted in a more vigorous response than was observed in units that had not been thinned previously.

Diversity

Fuel reduction treatments did relatively little to alter the allocation of aboveground cover of undergrowth species, while changing the number of species present. Reductions in species richness in the two treatments that included burning were the only statistically significant changes in diversity values captured by Shannon's index of diversity (H′). Changes in values for Pielou's index of evenness (J′) are linked to changes in H′ values, though often J′ does not respond as strongly (Shafi and Yarranton 1972; Smith and Wilson 1996). This was evident in the results of this study as changes in J′ values effectively mirrored changes in H′ values, presumably also in response to changes in richness.

One possible explanation for the decrease in diversity in the burn treatments could be the amount of time that had elapsed since treatment. Post-treatment measurements were taken the first growing season after fall burning. While many ecologists report that species diversity is highest immediately after disturbance (Ahlgren 1960; Conway 1981; Abrams and Dickman 1982; McGee *et al.* 1995; Grant and Loneragan 2001), the opposite has often been observed.

Many researchers have reported that diversity typically does not peak until several growing seasons after the disturbance; instead, disturbance events often lower diversity in the short term (Nieppola 1992;Collins *et al.* 1995). For example, Scherer *et al.* (2000) found that timber harvesting in the mixed-conifer forests of eastern Washington had little effect on species diversity three years after harvest, though diversity was reduced up until that time.

While disturbance creates the conditions for increased diversity, there are many factors which may not allow that to happen (Collins, Glenn, and Gibson 1995). This short-term negative influence of disturbance on diversity of the undergrowth has been explained by intra-specific competition. Rhizomatous or vegetatively-reproducing species can respond quickly to light disturbance and exclude seed reproducing species (Stickney 1986; Grant and Loneragan 2001). This is particularly true in Thin-only treatments if the soil is not disturbed (Dyrness 1973). Even under more extreme conditions such as a severe burn, vegetative reproducers can dominate the immediate postfire vegetation and reduce species richness (Turner *et. al*. 1997).

Another plausible explanation for decreased diversity in response to burning could be weather patterns. In 2001, total precipitation was 20 percent below the 30-year annual average (NCDC 2002). This dearth of moisture could have prevented the germination of species which otherwise might have colonized the burned units.

Vegetative characteristics

Vegetative characteristics represent the net consequence of vegetative change. Alterations to wildlife habitat and forage are borne out in the actual cover and frequency of the vegetation, particularly when specific species are considered. Intuitively, cover and frequency of the undergrowth should be correlated. However, these two measures of species abundance differed somewhat in their response to treatments.

Cover

Reduced graminoid and shrub cover in response to burn treatments, particularly relative to Control cover levels, was not consistent with many other reports in the literature. Other researchers have found that fire tends to increase undergrowth cover within the first year, particularly of graminoids (Harris and Covington 1983; Covington *et al.* 1997; Busse *et al.* 2000). A lack of response, or even a decrease in grass and shrub cover in the first year after disturbance, has been observed elsewhere (Gruel *et al.* 1986; Ayers *et al.* 1999). In both of the previous instances, however, grass and shrub cover in succeeding years exceeded the pre-burn condition, suggesting that future measurements may indicate a reversal of the current observed trend.

In contrast to the modest response from graminoids and shrubs, forb cover tended to increase in response to Burn-only treatments. Forb cover was still extremely low (3 percent), however, so the post-fire forb-dominated stage suggested by Abrams and Dickman (1982) and Stickney (1986) was not strongly evident. Furthermore, this modest increase in forb dominance may be short-lived as graminoids and shrubs recover from possible negative effects of burn treatments.

Thinning alone had some effect on the cover of lifeforms. In all cases, the remeasurement two years after treatment (2000) recorded approximately a 50 percent reduction in cover relative to pretreatment levels. This result was unexpected, as previous research has shown dramatic increases in cover, particularly of graminoids, within two years of thinning (Dyrness 1973; Bedunah *et al.* 1988). Somewhat in support of this observed response, however, McConnell and Smith (1965) noted that the three-year response to geometric thinning of ponderosa pine (*Pinus ponderosa*) stands in eastern Washington resulted in a relatively small, though significant increase in forage production. Additionally, it is possible that the reported decrease in cover was due to observation error; the field crew in 2000 could have consistently underestimated cover relative to estimates made in 1998 and 2001. Unfortunately, measurements were not taken in the unthinned units for 2000, precluding comparison with the controls.

Cover levels recovered significantly between 2000 and 2001 for all lifeforms in the Thin-only units. Both graminoids and shrubs increased appreciably over pretreatment levels. Forb cover simply returned to pre-treatment levels in 2001. These results were consistent with other reports in the literature (McConnell and Smith 1970; Dyrness 1973; Bedunah *et al.* 1988), although the modest graminoid response was somewhat unusual. Continued increases in cover are expected, based on studies indicating that peak response to thinning is observed 11-30 years after treatment in lodgepole pine (*Pinus contorta*) forests (Conway 1981), and more than eight years post-treatment in ponderosa pine forests (McConnell and Smith 1970).

Cover response to treatments also can be explained, in part, by the management history of the sites. All of the research areas had been partially harvested previously;

consequently, only minor differences were recorded in pre-treatment and post-treatment Stand Density Index (SDI). Pre-treatment stocking levels of the overstory ranged from 32 to 43% of maximum, a relatively open forest structure (Appendix 2). Thus thinning treatments only lowered SDI by about 35 percent. Given the modest density reduction in the overstory, a dramatic undergrowth response may not be expected.

Graminoid interactions

Many studies investigating undergrowth response to treatments focus solely on lifeform, and the principal components analysis (PCA) suggested that this was where the majority of effects were to be observed. In addition, however, there were some inter-specific interactions occurring among the graminoids, most of which appeared to reflect fire adaptations of individual species. Life history characteristics such as growth phenology and mechanism of reproduction could strongly influence species response to treatments.

The variable constructed to investigate these interactions consisted of the combined covers of pinegrass and Idaho fescue (*Festuca idahoensis*) contrasted against the cover of elk sedge. PCA analysis of the data suggested that burning increased the dominance of elk sedge relative to that of pinegrass and Idaho fescue. This response was expected, as Idaho fescue is notoriously fire sensitive (Gruell *et al.* 1982; Smith and Arno 1999; Busse *et al.* 2000). Pinegrass, in contrast, has often been shown to increase dominance of cover with burning treatments, at the expense of elk sedge and Idaho fescue (Bedunah *et. al*. 1988; Smith and Arno 1999).

Investigation of the unadjusted cover differences between 1998 and 2001 (Appendix 6) revealed that pinegrass had in fact decreased somewhat in response to burn treatments as well. Cover of Idaho fescue, meanwhile, declined dramatically in response to burn treatments. The constructed variable combined the slight reduction in pinegrass with the substantial decline in Idaho fescue to highlight the noticeable increase in the proportion of cover belonging to elk sedge.

The response of elk sedge to burn treatments is possibly correlated to burn intensity. While high intensity burns severely reduce cover of elk sedge, making way for greater increases of pinegrass, less intense burns tend to damage elk sedge less (Smith and Arno 1999). Light burns have been shown to stimulate growth of rhizomatous species, which more intense burns usually set back, to the detriment of species dependent on severe fire to free up resources (Ohmann and Grigal 1981; Grant and Loneragan 2001). Both pinegrass and elk sedge reproduce primarily through rhizomes (Johnson 1998), so both potentially have the opportunity to greatly increase cover from reproductive organs left intact by relatively light fires. In this instance, elk sedge resprouted within a month of the burn treatments (Youngblood 2002). The sexually reproductive pinegrass, however, may have had more limited opportunity to increase cover in the first season after burning.

Thin-only treatments did not elicit the production of florets from pinegrass or the reduced cover of Idaho fescue observed in the burned units of this study. This could have increased competition on elk sedge and resulted in the observed reduction in elk sedge cover (Appendix 6). In the third year after thinning, elk sedge dominance was reduced relative to pinegrass and Idaho fescue. This was inconsistent with McConnell and Smith's (1965) findings that pinegrass made up 78 percent of the increase in graminoid production in response to geometric thinning of ponderosa pine in eastern Washington.

Frequency

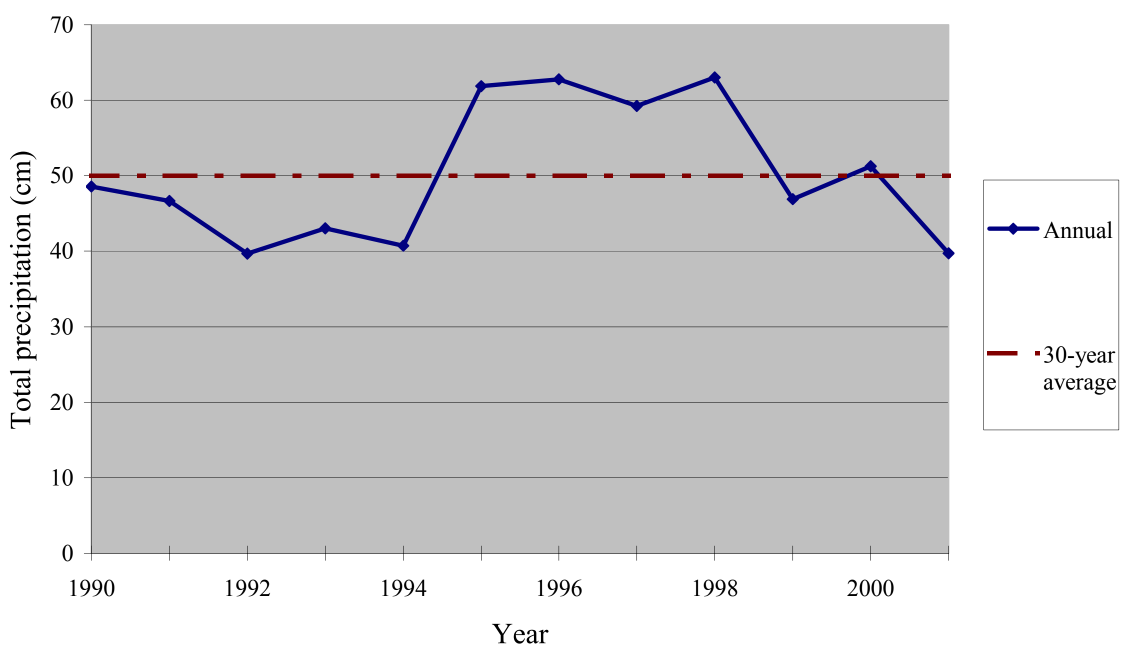
Changes in frequency could potentially have profound consequences for undergrowth species. While percent cover increases could represent a one-time flush of a particularly showy individual plant, frequency more accurately represents the distribution of individuals across the landscape. An event such as fire can remove the aboveground parts of a plant thus making them difficult to see and suggesting zero cover. Vegetative reproductive structures located in the ground may survive and even though the cover of such an individual has become zero, it will still play a role in the future. Measuring frequency can thereby provide insight as to future undergrowth trends.

Most of the 29 primary undergrowth species identified throughout the entire study did not decrease in frequency (Appendix 7). After three years, the majority of undergrowth species had either not changed, or were more prevalent. An increasing trend most effectively described changes in frequency, regardless of treatment. This was illustrated by observing the response of western yarrow (*Achillea millefolium*) and elk sedge from 1998 to 2001.

Two reasonable explanations exist for the pervasive increases in frequency. One reason for this trend could have been year-to-year changes in observers throughout the course of the study. This could have been driven by a shift in emphasis from simply identifying presence or abundance of certain species for habitat-typing purposes, to accurately identifying composition of the undergrowth vegetation.

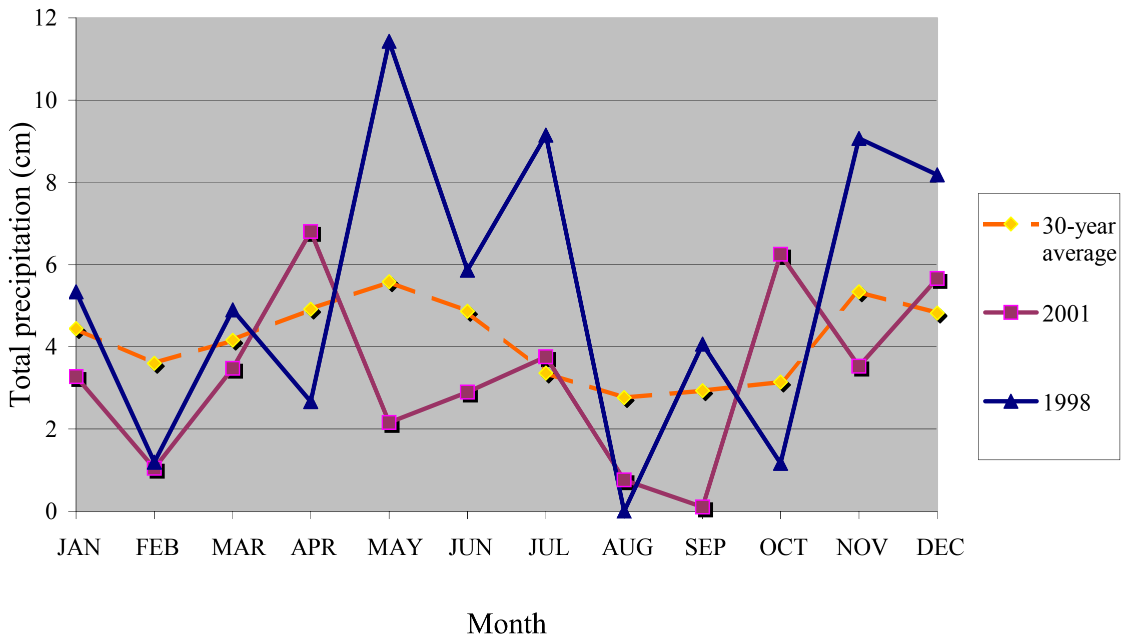
Western yarrow, elk sedge, and prairie Junegrass (*Koelaria macrantha*) were the species that best represented the trend of increasing frequency, regardless of treatment. Disturbance is known to increase the frequency of western yarrow, so the increased frequency of this species in the treated areas was expected. Both western yarrow and elk sedge are quite drought hardy as well (Johnson 1998; Kershaw *et al.* 1998). Prairie Junegrass is tolerant of disturbance (Kershaw *et al.* 1998), fairly drought hardy, and a colonizer into drought-stressed grasslands (Weaver and Albertson 1944). Increased frequency of these species in untreated units may be explained by precipitation in 2001 which was 20 percent lower than the 30-year average (Figure 12) (NCDC 2002). Such an event could increase the fitness of drought-hardy plants such as western yarrow, elk sedge, and prairie Junegrass.

Figure 12: Annual precipitation for the period 1990-2001 with the 30 year average annual precipitation for reference.



In addition to reduced overall precipitation in 2001, there was a substantial difference in the seasonality of precipitation between 1998 and 2001 (Figure 13) (NCDC 2002). For the months of May, June, and July, precipitation in 2001 was approximately half that in 1998, which could do even more to favor drought-hardy plants such as western yarrow, elk sedge, and prairie Junegrass. Western fescue (*Festuca occidentalis*) and smooth wildrye (*Elymus glaucus*) disappeared from the study units between 1998 and 2001 (Appendix 7). While reasonably drought resistant, these two grass species are often associated with moister sites (Hitchcock and Cronquist 1973; Kershaw *et al.* 1998; Johnson 1998), suggesting that water stress could have been an important contributor to their absence in 2001.

Figure 13: Total monthly precipitation for the 2001 and 1998 with the 30 year monthly average precipitation for reference.



Though there was a strong tendency toward increased frequency, or no response, there was still considerable variation in undergrowth frequency response to treatments. Four undergrowth species represented most of the variability in frequency among treatments: western yarrow, elk sedge, Idaho fescue, and prairie Junegrass. Western yarrow frequency increased regardless of treatment. Elk sedge and prairie Junegrass increased in frequency, particularly in units that received prescribed burning. Intense heat has been known to kill elk sedge (Smith and Arno 1999), suggesting that burning was low intensity, allowing the reproductive rhizomes to survive and capitalize on the disturbance.

Idaho fescue was not particularly stimulated by fuel reduction treatments, although low thinning did appear to increase the frequency of this species. Prescribed burning reduced the frequency of Idaho fescue, a response reported elsewhere for this fire-sensitive species (Gruell *et al.* 1982; Smith and Arno 1999; Busse *et al.* 2000). While a reduction in frequency was noted, it was slight, suggesting once again that the prescribed fires were not intense.

Future measurements and analyses

Immediate consequences of burning treatments and three-year responses to low thinning have been documented in this analysis. Trends identified to date could change, making future remeasurements very desirable. Long-term trends and their implications for land managers could thereby be assessed and quantified. This analysis identified several factors or modifications that could make such remeasurements more productive and useful.

The most drastic remeasurement design change would be to sample the vegetation using more but smaller plots, possibly nested within the existing plots. Doing so would provide a more descriptive representation of frequency. The current plot sizes gave some idea of the frequency of the more moderately distributed undergrowth, but very prevalent or rare vegetation was not precisely quantified.

Variability due to measurement error should be minimized. Percent cover is inherently difficult to estimate consistently between years. At least one person familiar with previous measurements should calibrate new field crews. Several recalibrations throughout the course of the field season may also be advisable. Measuring only the thinned units in 2000 served to reduce the amount of labor and cost in that year, but the value of the data collected in 2000 was consequently marginalized. All of the treatment units should be assessed in each remeasurement period. Otherwise, there is no way to account for annual variation due to weather and composition of field crews. Continuing to monitor the sites in June and July would also help reduce the effect of differences between years, especially if the units are measured in random order.

The techniques utilized to analyze this data set were fairly effective, though a few things could be changed. A less biased estimate of species evenness could be useful, in order to determine if the distribution of above ground cover is really changing in response to treatments, and not just because there are more species present. One alternative measure of evenness is E_{var} (Camargo 1993), which, according to Smith and Wilson (1996) is more equally sensitive to minor and abundant species, and is independent of species richness. This may be the preferred index, even though it is less common in the literature than Pielou's J′.

Accounting for variability between treatment units with the general linear model was useful. Characterization of soil with a continuous random variable such as bulk density or soil texture could make this process even more effective. Precipitation data were quite helpful with interpreting treatment effects on vegetation. Incorporating these data into the general linear model might help explain additional variation in vegetative response.

Analyzing the data by lifeform and with numeric indexes of diversity is convenient because it is more interpretable than trying to account for all of the species present on the units. Investigating the behavior of the more responsive species is more instructive than simply looking at groups of species, whenever possible. Future analyses could therefore focus on invasive species or sensitive natives that are of particular interest.

Conclusion

Undergrowth vegetation response to four fuel reduction treatments (no treatment, prescribed burn, low thinning, and low thinning followed by prescribed burning) was investigated in this study. Vegetative response to these treatments was measured or estimated in terms of biodiversity, percent cover, and species frequency.

With the Control treatment as a baseline, treatment effects were more clearly isolated from intrinsic annual variability. Treatment effects on the biodiversity of the undergrowth were assessed by comparing treated units to the untreated units (Control) in 2001. Vegetative measures in the Control units did change from 1998 to 2001. These changes reflected undergrowth response to annual weather perturbations and variability in observations resulting from changing field crews.

Short-term response (9 months) to Burn-only treatments suggested that burning significantly reduced diversity. A trend of declining cover was observed for graminoids and shrubs, while a more positive response to burning was observed for forb cover. Burn-only treatments tended to favor elk sedge and prairie Junegrass over other species of graminoids. Species frequency changed little for most species, but some, including elk sedge, prairie Junegrass, and arrowleaf balsamroot, increased.

Thin-only treatments did not exhibit undergrowth species diversity that differed significantly from that of the Control units in 2001. Two years after thinning, cover of the undergrowth was reduced by half. In the third year post treatment, cover of many species returned to, or increased above, pretreatment levels. Graminoids (especially pinegrass and Idaho fescue) and shrubs tended to increase in cover, while forb cover was equivalent in 1998 and 2001. Most undergrowth species increased in frequency three years post-thinning, especially prairie Junegrass, Idaho fescue, and arrowleaf balsamroot.

When low thinning was followed by prescribed burning, diversity in the following growing season decreased relative to the control units. Graminoid cover did not decline as much as in the Burn-only treatment. Forb cover was somewhat reduced, but shrub cover increased slightly compared to pretreatment. Thin-and-Burn treatments elicited an increase in frequency of most undergrowth species, particularly prairie Junegrass and, to a lesser extent, elk sedge. Undergrowth vegetation responded similarly in all treatments which involved burning. Fire-sensitive species declined in frequency and cover in the Thin-and-Burn treatments, suggesting a more intense fire than in the Burn-only treatments.

Fuel reduction treatments did not strongly influence the undergrowth vegetation in this study, possibly due to the intensity of treatments. Factors such as disturbance history and intensity of treatment likely influenced the observed responses. Trends observed in this study were only short term, particularly in the burned treatments. With continued monitoring of these treated areas, a great deal of insight could be gained as to the long-term effects of fuels reduction treatments on the undergrowth vegetation in ponderosa pine/Douglas-fir forests.

Appendix 1: Principal component (PC) coefficients for the three primary response variables used to determine species and assemblages of species to investigate.

Species	Cover		Constancy		
	PC 1	**PC 2**	**PC 1**	**PC 2**	**PC 3**
Achillea millefolium	0.000	0.000	0.407	-0.533	-0.057
Pseudoroegneria spicata	-0.001	-0.004	-0.040	-0.043	-0.017
Alnus incana	0.000	0.000	0.000	0.000	0.000
Amelanchier alnifolia	0.000	0.000	0.037	0.037	0.115
Arctostaphylos uva-ursi	0.000	0.001	0.006	0.005	0.021
Arnica cordifolia	0.017	0.119	0.069	0.068	-0.009
Balsamorhiza sagittata	0.000	0.003	0.179	0.006	-0.381
Berberis repens	0.000	0.000	0.009	-0.032	0.147
Calamagrostis rubescens	0.771	-0.559	0.082	0.076	0.016
Carex geyeri	0.296	0.517	0.339	0.279	-0.017
Carex rossii	0.000	0.000	-0.001	0.004	0.006
Ceanothus velutinus	0.000	0.000	0.000	0.000	0.000
Danthonia unispicata	0.000	0.000	-0.002	-0.004	-0.001
Elymus glaucus	0.000	0.000	-0.010	0.031	0.082
Festuca idahoensis	-0.056	-0.516	-0.113	-0.205	-0.031
Festuca occidentalis	0.000	0.000	0.001	0.008	-0.002
Koelaria macrantha	0.007	-0.021	0.168	-0.268	0.155
Linnaea borealis	0.000	0.000	0.002	0.005	0.013
Phleum pratensis	0.000	0.001	0.058	-0.031	0.167
Physocarpus malvaceus	0.001	0.017	0.012	0.007	0.056
Poa pratensis	0.002	0.001	0.026	0.020	0.186
Prunus virginiana	0.000	0.000	0.000	0.000	0.000
Ribes cereum	0.000	0.000	-0.003	-0.002	-0.002
Salix scouleriana	0.000	0.000	0.001	0.000	0.002
Shepherdia canadensis	0.000	0.000	0.000	0.000	0.000
Spirea betulifolia	0.007	0.019	0.109	0.061	-0.121
Stipa occidentalis	0.000	0.000	0.060	0.027	0.276
Symphoricarpos albus	0.103	0.102	0.047	0.004	-0.004
Vaccinium globulare	0.000	0.000	0.003	0.005	0.006

Extraction Method: Principal Component Analysis.
Coefficients are standardized.

Appendix 2: Overstory characteristics: overstory trees per hectare (Trees/ha), basal area (BA), percent Stand Density Index (SDI%), and overstory cover by unit and year for trees greater than 10 cm DBH.

Year	**Treatment**	**Unit**	**Trees/ha**	**Standard Error**	**BA (m^2/ha)**	**Standard Error**	**Percent SDI**	**Standard Error**	**Cover (%)**	**Standard Error**
1998	Control	15	225.0	34.7	19.8	4.3	39.8	7.9	62.0	6.0
1998	Control	18	378.9	74.2	18.5	2.2	37.6	4.7	62.1	5.7
1998	Control	23	227.8	25.5	17.7	1.6	33.4	3.1	63.6	5.5
1998	Control	245	315.0	43.5	14.5	1.7	31.3	3.6	54.3	7.6
1998	Burn	10B	225.0	29.5	14.4	2.1	30.3	4.2	44.8	7.5
1998	Burn	21	258.6	31.3	19.9	2.2	39.7	4.4	54.7	5.9
1998	Burn	24	231.6	39.2	14.9	2.1	30.1	4.2	41.7	7.5
1998	Burn	8B	282.6	25.8	17.0	2.0	36.0	4.2	54.8	6.8
1998	Thin	22	339.6	48.1	19.2	2.3	39.5	4.7	50.0	6.3
1998	Thin	6A	540.4	58.0	23.5	2.1	49.5	4.1	75.4	4.6
1998	Thin	7	556.0	73.1	28.9	3.0	60.4	6.4	76.0	5.4
1998	Thin	9	268.2	38.9	15.1	2.4	31.6	5.0	52.2	7.5
1998	Thin and burn	10A	250.0	42.7	13.2	1.5	27.7	3.2	40.8	5.8
1998	Thin and burn	1112	296.0	32.4	17.2	1.8	36.3	3.7	46.7	6.1
1998	Thin and burn	6B	469.0	41.1	18.7	1.4	40.7	3.0	55.9	4.4
1998	Thin and burn	8A	290.9	35.6	19.1	2.5	39.7	5.1	47.8	6.6
2000	Thin	22	186.1	26.4	12.9	1.2	25.7	2.5	49.3	7.1
2000	Thin	6A	258.7	25.4	15.0	0.8	30.1	1.4	72.3	5.3
2000	Thin	7	239.0	20.2	15.4	0.8	31.4	1.6	70.4	6.4
2000	Thin	9	204.3	19.3	13.8	1.2	28.4	2.3	49.6	8.0
2000	Thin and burn	10A	137.5	16.5	9.0	0.9	18.4	1.8	37.5	7.6
2000	Thin and burn	1112	120.0	8.7	9.3	0.9	18.9	1.7	45.2	6.1
2000	Thin and burn	6B	255.4	19.3	14.2	0.8	29.6	1.7	53.8	6.0
2000	Thin and burn	8A	163.6	11.2	12.3	1.3	25.0	2.4	48.7	7.2
2001	Control	15	227.5	22.9	20.0	2.5	40.2	4.6	70.0	9.2
2001	Control	18	376.3	51.4	24.0	1.6	46.1	3.4	67.4	8.8
2001	Control	23	249.1	24.7	19.7	1.6	37.2	3.1	76.4	6.1
2001	Control	245	301.2	42.3	15.9	1.9	33.6	3.9	58.1	7.9
2001	Burn	10B	188.8	26.0	12.7	1.8	26.5	3.7	39.0	8.6
2001	Burn	21	229.2	19.1	18.9	1.3	37.2	2.5	62.7	7.0
2001	Burn	24	202.4	38.6	13.7	1.6	27.2	3.0	36.5	6.7
2001	Burn	8B	269.6	20.7	18.0	1.6	37.5	3.3	63.5	7.9
2001	Thin	22	195.4	29.1	13.8	1.4	27.5	2.9	53.6	7.5
2001	Thin	6A	256.7	28.8	15.5	0.9	30.9	1.6	82.3	5.4
2001	Thin	7	235.0	21.9	15.5	0.9	31.5	1.7	68.8	6.6
2001	Thin	9	193.5	19.9	13.1	1.2	27.0	2.4	49.6	8.2
2001	Thin and burn	10A	135.4	16.6	9.2	0.9	18.9	1.9	34.2	7.8
2001	Thin and burn	1112	115.6	7.5	9.1	0.9	18.4	1.6	35.6	6.4
2001	Thin and burn	6B	217.2	15.7	13.0	0.7	26.9	1.4	50.3	6.9
2001	Thin and burn	8A	161.4	11.9	12.1	1.3	24.5	2.5	36.5	7.1

Appendix 3: Understory characteristics: trees per hectare (Trees/ha), basal area (BA), Stand Density Index (SDI) by unit and year for saplings: 0<DBH<10cm.

Year	Treatment	Unit	Trees/ha	Standard Error	BA (m^2/ha)	Standard Error	Percent SDI	Standard Error
1998	Control	15	340.0	128.2	0.5	0.2	1.9	0.9
1998	Control	18	184.2	47.3	0.5	0.1	1.8	0.5
1998	Control	23	219.6	137.8	0.2	0.2	0.7	0.6
1998	Control	245	138.1	44.8	0.3	0.1	0.8	0.3
1998	Burn	10B	26.2	8.9	0.0	0.0	0.1	0.1
1998	Burn	21	10.0	5.6	0.0	0.0	0.0	0.0
1998	Burn	24	176.1	86.6	0.3	0.1	0.9	0.3
1998	Burn	8B	2.2	2.2	0.0	0.0	0.0	0.0
1998	Thin	22	78.6	21.7	0.3	0.1	0.9	0.3
1998	Thin	6A	263.5	90.8	0.7	0.3	2.7	1.0
1998	Thin	7	274.0	50.0	0.8	0.2	2.5	0.5
1998	Thin	9	443.5	168.8	0.5	0.2	1.7	0.6
1998	Thin and burn	10A	93.8	36.9	0.3	0.1	0.8	0.3
1998	Thin and burn	1112	13.0	5.1	0.0	0.0	0.1	0.0
1998	Thin and burn	6B	219.0	53.0	0.6	0.1	2.0	0.4
1998	Thin and burn	8A	6.5	3.6	0.0	0.0	0.0	0.0
2000	Thin	22	75.0	18.3	0.2	0.1	0.8	0.2
2000	Thin	6A	343.3	149.0	0.8	0.3	2.8	1.1
2000	Thin	7	193.0	38.4	0.5	0.1	1.5	0.3
2000	Thin	9	307.6	115.5	0.4	0.2	1.3	0.5
2000	Thin and burn	10A	53.1	16.3	0.1	0.1	0.4	0.1
2000	Thin and burn	1112	11.1	5.1	0.0	0.0	0.1	0.0
2000	Thin and burn	6B	160.3	36.5	0.4	0.1	1.4	0.3
2000	Thin and burn	8A	8.7	3.0	0.0	0.0	0.1	0.0
2001	Control	15	390.0	106.7	0.5	0.2	2.1	0.8
2001	Control	18	171.1	36.9	0.4	0.1	1.6	0.5
2001	Control	23	355.4	173.5	0.3	0.2	0.9	0.6
2001	Control	245	159.5	41.5	0.3	0.1	1.0	0.3
2001	Burn	10B	9.5	3.2	0.0	0.0	0.1	0.0
2001	Burn	21	18.3	6.1	0.0	0.0	0.1	0.0
2001	Burn	24	118.5	82.9	0.3	0.2	0.9	0.5
2001	Burn	8B	3.3	1.8	0.0	0.0	0.0	0.0
2001	Thin	22	69.6	18.1	0.2	0.1	0.8	0.2
2001	Thin	6A	282.7	118.6	0.6	0.3	2.4	1.0
2001	Thin	7	181.0	40.6	0.4	0.1	1.3	0.3
2001	Thin	9	301.1	113.1	0.4	0.2	1.3	0.6
2001	Thin and burn	6B	20.8	7.6	0.1	0.0	0.2	0.1
2001	Thin and burn	8A	4.6	2.3	0.0	0.0	0.1	0.0
2001	Thin and burn	10A	30.2	8.9	0.1	0.0	0.4	0.1
2001	Thin and burn	1112	6.5	2.8	0.0	0.0	0.1	0.0

Appendix 4: Conifer regeneration characteristics: trees per hectare (Trees/ha), basal area (BA), Stand Density Index (SDI) by unit and year for seedlings: DBH=0.0cm.

Year	Treatment	Unit	Trees/ha	Standard Error	BA (m^2/ha)	Standard Error	Percent SDI	Standard Error
1998	Control	15	260.0	89.3	-	-	-	-
1998	Control	18	223.7	97.5	-	-	-	-
1998	Control	23	673.2	266.9	-	-	-	-
1998	Control	245	514.3	245.6	-	-	-	-
1998	Burn	10B	11.9	4.8	-	-	-	-
1998	Burn	21	25.0	10.9	-	-	-	-
1998	Burn	24	41.3	13.6	-	-	-	-
1998	Burn	8B	19.6	13.6	-	-	-	-
1998	Thin	22	82.1	28.7	-	-	-	-
1998	Thin	6A	505.8	214.9	-	-	-	-
1998	Thin	7	348.0	86.6	-	-	-	-
1998	Thin	9	513.0	229.5	-	-	-	-
1998	Thin and burn	10A	177.1	94.6	-	-	-	-
1998	Thin and burn	1112	44.4	28.9	-	-	-	-
1998	Thin and burn	6B	86.2	32.9	-	-	-	-
1998	Thin and burn	8A	0.0	0.0	-	-	-	-
2000	Thin	22	227.7	92.5	-	-	-	-
2000	Thin	6A	402.9	88.5	-	-	-	-
2000	Thin	7	255.0	69.6	-	-	-	-
2000	Thin	9	429.3	116.0	-	-	-	-
2000	Thin and burn	10A	186.5	102.8	-	-	-	-
2000	Thin and burn	1112	72.2	39.1	-	-	-	-
2000	Thin and burn	6B	62.1	19.2	-	-	-	-
2000	Thin and burn	8A	5.4	3.8	-	-	-	-
2001	Control	15	1030.0	343.6	-	-	-	-
2001	Control	18	394.7	204.7	-	-	-	-
2001	Control	23	2937.5	670.5	-	-	-	-
2001	Control	245	4836.9	1611.7	-	-	-	-
2001	Burn	10B	7.1	6.0	-	-	-	-
2001	Burn	21	30.0	18.0	-	-	-	-
2001	Burn	24	45.7	19.7	-	-	-	-
2001	Burn	8B	13.0	13.0	-	-	-	-
2001	Thin	22	553.6	216.7	-	-	-	-
2001	Thin	6A	697.1	182.7	-	-	-	-
2001	Thin	7	778.0	206.7	-	-	-	-
2001	Thin	9	2579.3	703.9	-	-	-	-
2001	Thin and burn	10A	29.2	15.2	-	-	-	-
2001	Thin and burn	1112	2.8	2.0	-	-	-	-
2001	Thin and burn	6B	6.9	3.5	-	-	-	-
2001	Thin and burn	8A	140.2	80.4	-	-	-	-

Appendix 5: Conifer characteristics: trees per hectare (Trees/ha), basal area (BA), Stand Density Index (SDI) by treatment and year. Seedlings: DBH=0.00cm; Saplings: 0.01<DBH<10.0cm; Overstory: DBH>10.0cm.

Size	**Year**	**Treatment**	**Trees/ha**	**Standard Error**	**BA (m^2/ha)**	**Standard Error**	**Percent SDI**	**Standard Error**
Seedlings	1998	Control	444.3	107.6	-	-	-	-
Seedlings	1998	Burn	24.7	5.8	-	-	-	-
Seedlings	1998	Thin	352.5	79.5	-	-	-	-
Seedlings	1998	Thin and burn	77.2	25.5	-	-	-	-
Seedlings	2000	Thin	324.5	46.3	-	-	-	-
Seedlings	2000	Thin and burn	81.1	26.9	-	-	-	-
Seedlings	2001	Control	2408.2	476.3	-	-	-	-
Seedlings	2001	Burn	24.7	8.0	-	-	-	-
Seedlings	2001	Thin	1102.0	196.8	-	-	-	-
Seedlings	2001	Thin and burn	40.8	18.8	-	-	-	-
Saplings	1998	Control	219.9	54.4	0.3	0.1	1.3	0.3
Saplings	1998	Burn	51.0	21.6	0.1	0.0	0.3	0.1
Saplings	1998	Thin	255.9	47.6	0.6	0.1	1.9	0.3
Saplings	1998	Thin and burn	88.3	19.2	0.2	0.0	0.8	0.2
Saplings	2000	Thin	224.8	47.8	0.5	0.1	1.6	0.3
Saplings	2000	Thin and burn	62.4	12.6	0.2	0.0	0.5	0.1
Saplings	2001	Control	276.7	61.8	0.4	0.1	1.4	0.3
Saplings	2001	Burn	36.6	20.0	0.1	0.0	0.3	0.1
Saplings	2001	Thin	203.4	41.5	0.4	0.1	1.4	0.3
Saplings	2001	Thin and burn	16.0	3.3	0.1	0.0	0.2	0.0
Overstory	1998	Control	274.4	23.0	17.2	1.3	34.5	2.5
Overstory	1998	Burn	231.4	16.8	15.6	1.1	31.9	2.3
Overstory	1998	Thin	411.8	31.1	20.8	1.4	43.5	2.8
Overstory	1998	Thin and burn	317.0	21.7	16.2	1.0	34.4	2.0
Overstory	2000	Thin	219.9	12.0	14.2	0.5	28.6	1.1
Overstory	2000	Thin and burn	165.5	9.6	10.8	0.6	22.2	1.1
Overstory	2001	Control	281.3	18.6	19.6	1.0	38.5	1.9
Overstory	2001	Burn	217.5	13.6	15.7	0.8	31.8	1.7
Overstory	2001	Thin	218.4	13.1	14.4	0.6	29.0	1.1
Overstory	2001	Thin and burn	154.1	8.2	10.5	0.5	21.5	1.0

Appendix 6: Change in average cover between 1998 and 2001 by treatment, with associated standard error (SE).

Treatment	Control		Burn		Thin		Thin and burn		Total	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Achillea millefolium	0.87	0.15	0.73	0.06	0.56	0.11	0.55	0.10	0.68	0.06
Alnus incana	-0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Amelanchier alnifolia	-0.05	0.05	-0.03	0.02	0.04	0.02	-0.03	0.02	-0.02	0.02
Arnica cordifolia	6.27	5.24	1.33	2.32	-1.26	0.49	-0.38	1.22	1.49	1.51
Arctostaphylos uva-ursi	0.33	0.32	0.00	0.00	0.00	0.00	0.00	0.00	0.08	0.08
Balsamorhiza sagittata	0.13	0.11	0.32	0.26	0.45	0.25	0.36	0.31	0.31	0.11
Berberis repens	0.09	0.08	-0.15	0.13	0.16	0.09	-0.06	0.15	0.01	0.06
Carex geyeri	15.39	6.20	3.32	2.04	3.69	2.09	0.30	0.61	5.67	2.14
Carex rossii	-0.02	0.18	-0.01	0.01	-0.05	0.05	0.06	0.09	-0.01	0.05
Calamagrostis rubescens	12.19	4.60	3.10	4.18	5.80	1.79	-4.44	0.64	4.16	2.12
Ceanothus velutinus	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Danthonia unispicata	0.00	0.00	-0.32	0.22	0.00	0.00	-0.17	0.22	-0.12	0.08
Elymus glaucus	-0.41	0.31	-0.89	0.67	-0.02	0.02	0.00	0.00	-0.33	0.19
Festuca idahoensis	0.43	1.52	-10.03	3.66	0.48	1.07	-1.61	0.58	-2.68	1.45
Festuca occidentalis	-0.39	0.34	0.00	0.00	0.00	0.00	0.00	0.00	-0.10	0.09
Koelaria macrantha	0.02	0.12	1.55	2.02	0.28	0.11	3.75	1.38	1.40	0.67
Linnaea borealis	-0.06	0.34	0.00	0.00	0.00	0.00	0.00	0.00	-0.01	0.08
Physocarpus malvaceus	0.91	0.91	-0.49	0.46	0.03	0.03	-0.15	0.15	0.07	0.27
Phleum pratensis	-0.20	0.27	-0.17	0.18	0.11	0.08	0.15	0.12	-0.03	0.09
Poa pratensis	-1.56	1.49	-3.96	1.86	-2.40	0.99	-1.62	1.24	-2.38	0.69
Prunus virginiana	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pseudoroegneria spicata	-0.63	0.89	-0.28	0.24	-0.32	0.22	-0.24	0.12	-0.37	0.22
Ribes cereum	0.00	0.00	-0.08	0.08	0.00	0.00	-0.01	0.01	-0.02	0.02
Salix scouleriana	-0.02	0.06	-0.03	0.03	-0.01	0.02	0.00	0.00	-0.01	0.02
Shepherdia canadensis	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Spirea betulifolia	2.11	0.47	0.24	0.18	2.11	0.63	0.63	0.36	1.27	0.30
Stipa occidentalis	-0.56	0.42	-0.09	0.06	-0.43	0.49	0.01	0.00	-0.27	0.16
Symphoricarpos albus	7.99	3.04	0.66	0.09	3.82	2.59	1.15	0.78	3.40	1.18
Vaccinium globulare	-0.10	0.13	0.00	0.00	0.02	0.03	0.00	0.00	-0.02	0.03

Appendix 7: Change in frequency between 1998 and 2001 by treatment with associated standard error (SE).

Treatment	Control		Burn		Thin		Thin and burn		Total	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Achillea millefolium	0.98	0.03	0.91	0.03	0.80	0.10	0.98	0.01	0.92	0.03
Alnus incana	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Amelanchier alnifolia	0.14	0.12	-0.01	0.06	0.11	0.04	-0.04	0.03	0.05	0.04
Arnica cordifolia	0.21	0.05	0.09	0.07	0.05	0.06	-0.09	0.07	0.06	0.04
Arctostaphylos uva-ursi	0.08	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.01
Balsamorhiza sagittata	0.17	0.11	0.29	0.11	0.42	0.21	0.31	0.15	0.30	0.07
Berberis repens	0.20	0.14	-0.01	0.03	0.06	0.04	0.10	0.03	0.08	0.04
Carex geyeri	0.17	0.06	0.18	0.08	0.18	0.13	0.16	0.11	0.17	0.04
Carex rossii	-0.06	0.12	-0.01	0.01	-0.02	0.02	-0.01	0.05	-0.02	0.03
Calamagrostis rubescens	0.11	0.06	0.09	0.04	0.03	0.09	0.03	0.07	0.06	0.03
Ceanothus velutinus	0.00	0.00	0.00	0.00	0.00	0.00	-0.01	0.01	0.00	0.00
Danthonia unispicata	0.00	0.00	-0.01	0.04	0.00	0.00	0.03	0.02	0.01	0.01
Elymus glaucus	-0.25	0.14	-0.28	0.19	-0.07	0.07	0.00	0.00	-0.15	0.06
Festuca idahoensis	0.05	0.03	-0.01	0.09	0.22	0.12	-0.06	0.12	0.05	0.05
Festuca occidentalis	-0.15	0.13	-0.01	0.01	0.00	0.00	0.00	0.00	-0.04	0.03
Koelaria macrantha	0.27	0.17	0.37	0.09	0.47	0.10	0.91	0.01	0.50	0.08
Linnaea borealis	-0.05	0.04	0.00	0.00	0.00	0.00	0.00	0.00	-0.01	0.01
Physocarpus malvaceus	0.06	0.03	0.01	0.01	0.00	0.00	0.01	0.02	0.02	0.01
Phleum pratensis	0.20	0.08	0.22	0.12	0.16	0.06	0.16	0.07	0.18	0.04
Poa pratensis	-0.07	0.17	-0.09	0.08	0.15	0.12	0.14	0.11	0.03	0.06
Prunus virginiana	0.01	0.01	-0.01	0.02	0.01	0.01	0.00	0.00	0.00	0.01
Pseudoroegneria spicata	0.03	0.03	-0.09	0.08	-0.11	0.11	0.04	0.03	-0.03	0.04
Ribes cerium	0.00	0.00	-0.05	0.05	0.00	0.00	-0.01	0.01	-0.01	0.01
Salix scouleriana	0.04	0.03	0.01	0.01	0.02	0.01	0.01	0.01	0.02	0.01
Shepherdia canadensis	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Spirea betulifolia	0.29	0.05	0.14	0.09	0.05	0.02	0.12	0.06	0.15	0.03
Stipa occidentalis	0.28	0.10	-0.02	0.03	0.09	0.10	0.02	0.01	0.09	0.04
Symphoricarpos albus	0.16	0.07	0.20	0.08	0.10	0.02	0.16	0.05	0.15	0.03
Vaccinium globulare	0.04	0.03	0.00	0.00	-0.02	0.01	0.00	0.00	0.01	0.01

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