

## The Effects of Spring Burning and Grass Seeding in Forest Clearcuts on Native Plants and Conifer Seedlings in Coastal Washington

### Abstract

Seeding clearcuts with grasses and legumes is used to increase ungulate forage, reduce browsing on seedlings, reduce competition with seedlings, or control erosion, but the effects on native plants and conifer seedlings are poorly known. Also, a shift to spring or no burning of clearcuts may alter the effectiveness of seeding and interactions with native species and conifer seedlings. I quantified over 5 yr the effects of burning (none, spring) and forage grass seeding (0, 22, or 44 kg/ha seed, plus fertilizer) on seeded forage production, native and exotic plant cover and richness, and the survival and growth of planted conifer seedlings on two clearcuts in coastal forest of western Washington. Burning doubled production of seeded grass, but seeding rate (22kg/ha vs. 44 kg/ha) had no effect. Grass biomass peaked in 3 yr, then declined to negligible levels. Detrimental effects of both burning and seeding on native species appeared small and short-lived over a 5-yr period. Native plant species richness decreased after burning, but after 5 yr nearly equaled that of unburned areas. Native plant species richness was unaffected by seeding rate. Native plant cover increased with time after clearcutting regardless of treatment. In burned areas, seeding reduced native cover slightly. Exotic species richness and cover were highest in burned areas. About 25% of the planted western hemlock seedlings died, mostly during the first year. Mortality was 64% higher in seeded, burned areas than in unseeded, burned areas. Final seedling height was 4% higher in the burned than unburned areas, and seeding rate had no effect on height. Results of this study likely would apply to similarly treated western hemlock plant associations in coastal forests.

### Introduction

Seeding forage grasses and legumes in recently burned clearcuts has been widely used in the western United States and Canada since the 1950s to increase the quantity of herbaceous forage for elk (*Cervus elaphus*), reduce ungulate browsing damage on conifer seedlings, suppress vegetation competing with conifer seedlings, or provide for erosion control (Ramsey and Krueger 1986, Owston et al. 1992, Powell et al. 1994, Becker et al. 1996). Seeding non-native grasses and herbs is also a widespread practice for erosion control and restoration of soil-nutrient cycles after wildfires (Klock et al. 1975, Tiedemann and Klock 1976, Agee 1993). Nevertheless, seeding programs are controversial (Owston et al. 1992).

Grasses compete with conifer seedlings, slowing growth or increasing mortality (Clark and McLean 1978, Crouch 1979, Dimock et al. 1983, Stewart et al. 1984, McDonald 1986, Bell et al. 2000), especially in moisture-limited sites (Clark and McLean 1978, Crouch 1979, Dimock et al. 1983). Seeding may increase herbivory by attracting ungulates (Crouch 1969, Hines and Land 1974,

Becker 1989), rather than decrease herbivory by providing a high-quality forage alternative to conifer seedlings (Campbell and Evans 1975). Herbivory on woody plants from microtine rodents that flourish in dense grass swards created by seeding and fertilization also can be a problem (Lawrence et al. 1961, Askham 1992). Seeding typically introduces exotic or weedy species and potentially alters successional development of native plant communities (Tiedemann and Klock 1976, Helvey and Fowler 1979, Owston et al. 1992, Agee 1993, and see Becker et al. 1996 for review).

Changes in post-harvest management practices in western U.S. forests also have led to questions about the impact and value of forage seeding in logged areas. Air quality standards have forced a shift from fall burning of logging debris for site preparation and fuel reduction (Owston et al. 1992) to no burning or burning during the spring, when fire intensity and fuel reduction are relatively light (Miller et al. 1993). Cool spring burns leave more woody debris and impact native plant communities less than hotter fall burns (Dyrness 1973, Halpern 1988, Halpern and Spies 1995). Consequently, seeded grass production may be low because of limited availability of germination sites and from increased competition with residual

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vegetation (Owston et al. 1992). Greater rates of seeding, however, might compensate for less germination space. Diminished, yet still potentially useful, amounts of forage in lightly burned areas might compete less with native species or planted conifers, especially in coastal areas with high rainfall. Not burning may reduce invasion of ruderal species (Isaac and Hopkins 1937, West and Chilcote 1968, Halpern et al. 1997) that are increasingly becoming an important issue for coastal forest management (DeFerri and Naiman 1994, Heckman 1999, Thysell and Carey 2000).

Information to assess the interactive effects of new burning practices and forage seeding is limited. Grass seeding programs have been poorly monitored, many have had no experimental replication or control, predictability of effects is low because of high environmental variability among sites, and many studies have not been published in the scientific literature (Agee 1993, Powell et al. 1994, Becker et al. 1996). The goal of this study was to quantify experimentally the short-term (5 yr) effects of spring burning and rate of grass seeding on production of seeded forage, native and exotic plant species groups, and the survival and growth of planted conifer seedlings in a coastal western hemlock (*Tsuga heterophylla*) forest in western Washington. Because there were two replicates in the study and many species were sparsely distributed, I analyzed treatment effects on native and exotic plant species as groups, not individual species. An appendix of species responses is available from the author. I applied fertilizer with seed on seeded areas (the same amount regardless of seeding rate), but not to unseeded areas, to mimic typical forest management practices (Owston et al 1992; Mark Ostwald, Olympic National Forest, Quinault Washington, personal communication.). Hence, the seeding treatment was designed to compare the effects of no seed + no fertilizer vs. two rates of seed application + a constant amount of fertilizer. I did not explicitly test the effects of burning and seeding treatments on ungulate herbivory because that would have required a more geographically extensive experimental design than was feasible (Becker et al. 1996). However, I did measure and report the occurrence of small and large mammal herbivory on conifer seedling mortality.

I tested two hypotheses: 1) Spring burning, compared to no burning, will: (a) enhance the production of seeded forage; (b) reduce the short-

term species richness and cover of native plants; (c) increase the species richness and cover of exotic species; and (d) enhance the survival and growth of planted conifer seedlings; 2) Increasing forage grass seeding rate will: (a) increase forage production; (b) decrease richness and cover of native plants; (c) increase the richness and cover of exotic species; and (d) decrease survival and growth of planted conifer seedlings.

## Methods

### Study Area

The experiment was in two clearcuts (15.0 ha and 9.3 ha) of the Bull timber sale, Olympic National Forest, located in the lower Matheny Creek drainage on the western Olympic Peninsula (latitude 47° 33'N; longitude 123° 57'W). The units were separated by a forested buffer of about 200 m. Forests at both sites were clearcut in January 1990 and yarded in March and April 1990. Experimental units were broadcast burned with a helitorch under clear weather during late May 1991, seeded with forage grasses in fall 1991, then planted with conifer seedlings during mid-February 1992. Seedlings were planted at a 3.3-m spacing, with 15 cm scalping of unburned sites only. Planting stock was a typical mixture of western hemlock plug 1-0 (50%), western redcedar (*Thuja plicata*) plug 1-0 (25%), and Douglas-fir (*Pseudotsuga menziesii*) bare-root 2-0 (25%) seedlings. Only western hemlock seedlings were sufficiently abundant in all plots (mean 31 seedlings, range 15-57) for measurement of treatment effects on seedling growth and survival. No seedlings were replanted. Data were collected during the fall from 1992 through 1996.

Forest vegetation in the study area is in the western hemlock series, with adjacent upslope areas in the Pacific silver fir (*Abies amabilis*) series. The dominant plant association is western hemlock/Alaska huckleberry-salal (*Tsuga heterophylla/Vaccinium alaskaense-Gaultheria shallon*) (Henderson et al. 1989). The topography is gently sloping (<10°) with a northerly aspect. Soils are highly productive, medium and moderately coarse-textured, moderately-deep till soils in the Mudcreek-Kalaloch complex typical of level remnant outwash terraces (Olympic National Forest 2000). Elevation at the site is about 200 m. Mean annual precipitation at a nearby weather station during the study period (1990-1996) was 292 cm

(range 224-343 cm), and mean annual temperature was 10°C (range 9-11°C) (NOAA 1997). Mean annual precipitation at the site likely was somewhat higher, approximately 330 cm (Henderson et al. 1989).

### Experimental Design

I allocated burning and seeding treatments in a randomized complete block split-plot design, where each of the two clearcut units was a block (Federer 1955) (Figure 1). Burning was the whole-plot treatment. I randomly selected half of each

block for spring burning and the other half was to be left unburned. However, because of the vagaries of burning clearcuts, about two-thirds of each block was burned. Burning occurred during April 1991.

Within burned and unburned whole-plots, I randomly assigned three seeding rates (0 kg/ha, 22 kg/ha, and 44 kg/ha) to split-plots of approximately 40 x 40 m (Figure 1). Only one split-plot of each seeding treatment was physically possible in the unburned area, but two split-plots of each seeding treatment were randomly located in the

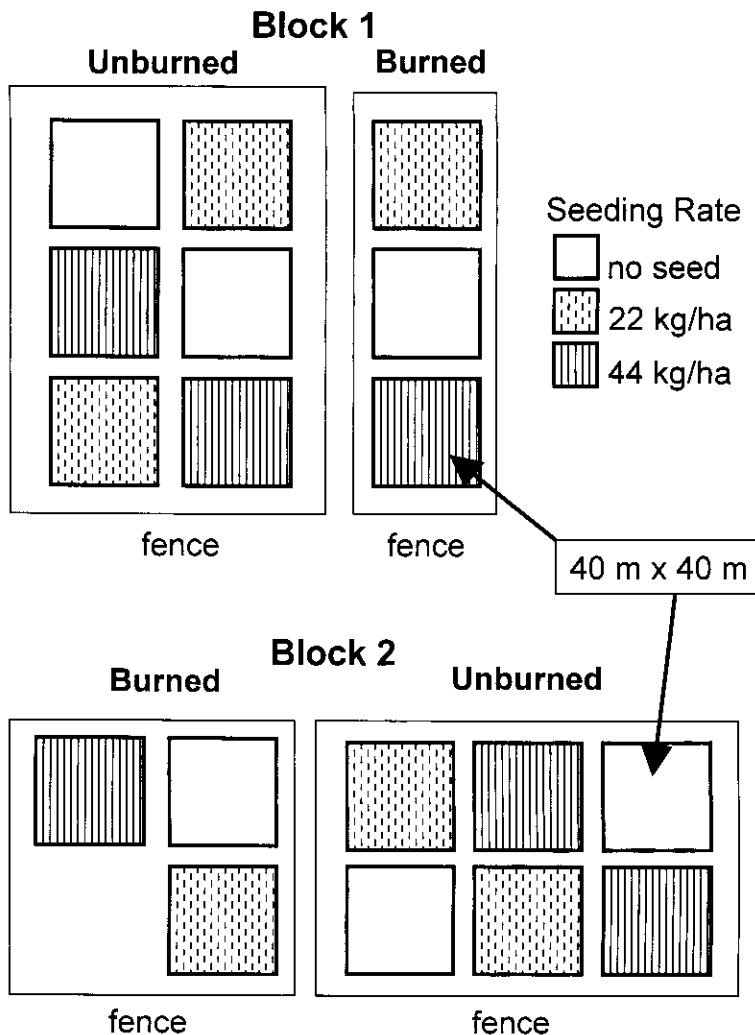


Figure 1. Experimental layout of whole-plot burn and split-plot seeding treatments to test the effects of spring burning and seeding rate on seeded grass production, native plant communities, and planted seedling mortality in the Matheny Creek drainage on the western Olympic Peninsula, Washington.

larger burned portion of each block. The unconventional use of two split-plots of each seeding treatment in burned areas was done to sample proportionately similar areas of the burned and unburned treatment areas. Thus, nine treatment split-plots were established in each of the two blocks. Details of the analysis of variance (ANOVA) model are in the Statistical Analysis section.

I applied seed and fertilizer during October 1991 with hand seeders to ensure accurate and homogeneous application. Following standard District management practices, I applied 16N-16P-16K elemental fertilizer only to seeded treatment plots at the rate of 336 kg/ha. The seed mix was 27% Potomac orchard grass (*Dactylis glomerata*), 21% white clover (*Trifolium repens*), 21% tetraploid perennial ryegrass (*Lolium perenne*), 21% birdsfoot trefoil (*Lotus corniculatus*), and 10% annual ryegrass (*Lolium multiflorum*) by weight. Germination rate certified by the seed company was 85% for orchard grass, 90% for annual ryegrass, 85% for perennial ryegrass, 82% for white clover, and 59% for birdsfoot trefoil. Legume seed was pre-inoculated with Dormal-brand nitrogen-fixing bacteria. The legumes, however, failed to germinate or became established at only trace levels. Seed and fertilizer also were applied selectively to areas outside and adjacent (within 50 m) to the experimental plots to lure deer and elk away from the experimental area.

The treatment area was enclosed with four strands of smooth wire electrified with a solar-powered charger to exclude black-tailed deer (*Odocoileus hemionus*) and elk. The fence functioned for about 1 yr, then the chargers failed and were not replaced. The fence remained standing, but did not totally exclude elk or deer. To estimate the potential effects of unplanned ungulate herbivory on dependent variables, I monitored elk and deer use by pellet-counts and incidence of browsing on planted conifer seedlings.

#### Field Measurements

National Forest personnel estimated pre- and post-burn amounts of woody debris in each block by the planar intersect method (Brown 1974), as implemented in standard Forest Service fuel inventories. Eight (Block 2) and 12 (Block 1) random debris plots were measured in blocks before burning, then remeasured after burning to esti-

mate mean debris reduction for the entire burned portion of each block.

I measured the treatment responses of biomass production for seeded grasses, species richness and cover of native and exotic species, and growth and mortality of planted conifer seedlings in the central 20 x 20 m area within each treatment split-plot. Native and exotic species designations were based on Hitchcock and Cronquist (1973). Plant species cover <1 m tall was estimated from forty 20 x 50 cm quadrats (Daubenmire 1959, 1968) in each treatment split-plot. I located quadrats in each treatment split-plot along four permanent 20-m transects systematically located 4 m apart.

I placed the first quadrat 1 m from the end of the tape, and each successive quadrat at 2-m intervals along the transect. Cover was measured each year at the same points on the left side of the transects. I estimated aerial cover by cover classes (Daubenmire 1959), which minimized annual variation in cover estimates among different field personnel. Mid-points of cover classes were used in the analysis. I recorded elk or deer pellets in cover quadrats to estimate use of the treatment areas. I used line-intercept estimates along the permanent 20-m transects to estimate cover of vegetation  $\geq 1$  m tall.

I estimated the annual above-ground production of seeded forage biomass during the last 2 wk of September by clipping seeded grass and legume biomass at ground level within sixteen 20 x 50 cm quadrats per seeded treatment split-plot. I clipped four quadrats along each of the permanent sample transect lines at 4-m intervals on the opposite (right) side from the cover quadrats and 1 m away from the transect. Each year the position of the first clip quadrat was shifted 1 m up the transect to avoid previously clipped areas. Live material and standing dead material from the current year's growth was clipped. Clipped biomass was bagged and dried at 100°C (Milner and Hughes 1968) for 48 hr to determine dry weight.

I tagged all western hemlock seedlings planted within treatment split-plots during 1992 and monitored their survival annually for three growing seasons through 1995. I randomly selected 20 western hemlock seedlings in each treatment split-plot at the beginning of the study to measure height (measured at the end of the leader). At the end of

the 5-yr sample period, each split-plot had an average 18 western hemlock seedlings (range 8-23 seedlings/split-plot) to estimate average height for the treatment. Vegetation within a 2-m radius of tagged trees was not disturbed by biomass clipping and other measurements. I used the field guide by Lawrence et al. (1961) to identify which animal species browsed western hemlock seedlings.

### Statistical Analysis

For each treatment split-plot, I calculated mean values of grouped native and exotic plant species cover, forage biomass, and conifer seedling survival and height from the individual quadrat or seedling measurements. Data for the two split-plot replicates of each seeding treatment within burned portions of blocks were averaged, which resulted in equal replication of seeding treatments per block (three burned and three unburned seeding values) for conventional blocked split-plot ANOVA designs (Federer 1955). The alternative to averaging for a balanced design was to exclude data for ~35% of the experimental area, which was deemed less desirable than averaging (Tim Max, US Forest Service, PNW Research Station, Portland, Oregon, personal communication).

I graphically determined how well data for dependent variables met the assumptions for ANOVA by examining for homogeneity of variance among groups and normality of distribution. A square root transformation ( $\sqrt{x + 0.5}$ ) was applied to exotic species cover values, mortality and height data for western hemlock, and ungulate pellet frequencies to correct for uneven variance or skewed distributions.

I analyzed treatment effects on seeded grass production, native and exotic plant species cover and richness, seedling mortality, and ungulate use over the 5-yr study period by repeated measures ANOVA for a blocked split-plot design (Table 1). Seedling height and browsing occurrence were measured at the end of the experimental period, so I analyzed treatment effects on those variables without the repeated measures component in the ANOVA model. I did orthogonal contrasts to test elements of my hypotheses among levels of seeding, years, or cell means of the burning\*seeding, year\*burning, or year\*seeding interaction terms if the main ANOVA indicated significant effects. SYSTAT (SPSS 1997) software was used for parametric statistical analysis.

TABLE 1. Analysis of variance model for a repeated-measures, randomized complete block, split-plot experiment to evaluate the effects of spring burning and forage seeding rates.

Source of Variation	df
Whole-plots	
Replicate	1
Burn	1
Error(a)	1
Split-plots	
Seeding	2
Seeding * burning	2
Error (b)	4
Repeated measures (5 yr)	
Time	4
Burn * time	4
Seed * time	8
Burn * seed * time	8
Error (c)	24
Total	59

A significance level of  $P = 0.10$  was accepted for all statistical analyses. Although less conservative than  $P = 0.05$ , particularly with the relatively small sample size in this experiment (two replicate blocks), I considered  $P = 0.10$  to be an acceptable chance of error for forest management issues and well within the bounds of statistical convention (Zar 1999). A significant difference is implied where a difference among means is reported in the text. For results presented primarily in figures, exact  $P$ -values are provided in the text to allow readers to assess the probability of error relative to their own standard of significance (Zar 1999).

Treatment effects on native and exotic plant assemblages were examined by canonical correspondence analysis (CCA) with the CANOCO program (Jongman et al. 1987, ter Braak 1988). Dependent variables were the mid-point of cover-class values for individual species in each of the 18 split-plots. Nominal independent environmental variables were the burning and seeding treatments for each split-plot. Block identification was entered as a covariate to control among-block variation for analysis of field experiments with CCA (ter Braak 1988). Rare species were downweighted.

The strength of the CCA ordination was represented by the trace, eigenvalues of the axes, and intraset correlations (Jongman et al. 1987). The eigenvalue is a measure of separation of the species distributions along the ordination axis, or the

amount of variation explained by the axis and environmental variables. The trace is the sum of all eigenvalues in the ordination. Intrasets correlations measured the strength of association between the individual environmental variables and the ordination axes.

I tested the significance of the trace and the eigenvalue of the first axis by Monte Carlo permutation tests of 99 runs within the CANOCO program (ter Braak 1988). An observed trace and first eigenvalue significantly higher than calculated by the Monte Carlo permutations indicated significant multi-species differences in vegetation among the treatments.

I used biplots of species ordination scores and the centroids of the nominal environmental variable (treatment) scores (Jongman et al. 1987, ter Braak 1988) to display and interpret relationships of species groups to treatments. Treatment centroids represent the weighted average of treatment split-plot scores based on species composition. The locations of the species points and the treatment centroids jointly reflect the relative abundance of species, or collectively species groups, in relation to environmental gradients, in this case the treatments. Moreover, a location close to the origin indicates a relatively weaker relationship to the environmental gradient than a more distant location.

## Results

### Debris Reduction from Burning

Spring burning reduced total woody fuel by 19%, leaving from 92 to 142 Mg/ha debris in the two experimental blocks (Table 2). Fine ( $\leq 7.6$  cm diameter) and large woody debris decreased 34% and 15%. Fuel depth overall was reduced 57%,

to an average post-burn depth of 11 cm. An average 11 cm of duff remained after burning, which was an 18% to 29% reduction from initial conditions. Fuel moisture and fire behavior information were not collected by District fire staff who conducted the burn, which they characterized as a low-intensity, very good, early-spring prescribed burn (McCray 1991).

### Ungulate Use

Elk and deer use of treatment plots was low (annual range 0.6 to 0.9 pellet groups/split-plot) and did not vary over the study period. Averaged across the study period, ungulate use was unaffected by burning, seeding, or their interaction. However, ungulates did use burned areas differently among years ( $P = 0.10$ ) in apparent response to forage availability.

During the first year (1993) ungulates used unburned plots more heavily (1.25 pellet groups/plot) than burned plots (0.21 pellet groups/plot). Coincident with peak forage grass production in burned areas during the second and third years, ungulates switched to using burned areas (1994, 1.46 pellet groups/plot; 1995, 1.21 pellet groups/plot) while avoiding unburned areas (no pellets recorded). During the fourth year (1996), when grass production dropped markedly, use shifted away from burned areas (0.21 pellet groups/plot) back to unburned areas (1.67 pellet groups/plot).

### Forage Production

Burning more than doubled the annual production of seeded grass ( $P = 0.049$ ) (Figure 2). Peak production in 1994 averaged 977 kg/ha in burned areas, and 289 kg/ha in unburned areas. Grass production in both burned and unburned treatments was similar during the first 2 yr, increased

TABLE 2. Pre- and post-burn fuel characteristics in two experimental blocks for spring burning and grass seeding studies in the Bull timber sale, Matheny Creek, Washington during 1991. An asterisk indicates a significant change ( $P \leq 0.10$ ) between mean pre- and post-burn amounts based on paired t-tests.

Fuel Type	Mean				Block 1 (n = 12)				Block 2 (n = 8)			
	Un-burned	Pre-burn	Post-burn	Change	Un-burned	Pre-burn	Post-burn	Change	Un-burned	Pre-burn	Post-burn	Change
Fine fuel (Mg/ha $\leq 7.6$ cm)	32.4	32.0	21.1	-34%*	36.9	31.1	20.6	-34%	28.0	33.0	21.5	-35%
Large fuel (Mg/ha $> 7.6$ cm)	127.6	112.1	96.3	-15%*	71.3	87.4	71.7	-18%	183.9	136.8	120.9	-12%
Total fuel (Mg/ha)	160.0	144.1	117.4	-19%*	108.2	118.5	92.4	-22%	211.8	169.7	142.4	-16%
Duff depth (cm)	11.4	15.4	11.4	-24%	13.0	21.1	15.0	-29%	9.9	9.7	7.9	-18%
Fuel depth (cm)	29.0	25.7	11.2	-57%*	31.5	25.1	8.4	-67%	26.4	26.2	14.0	-47%

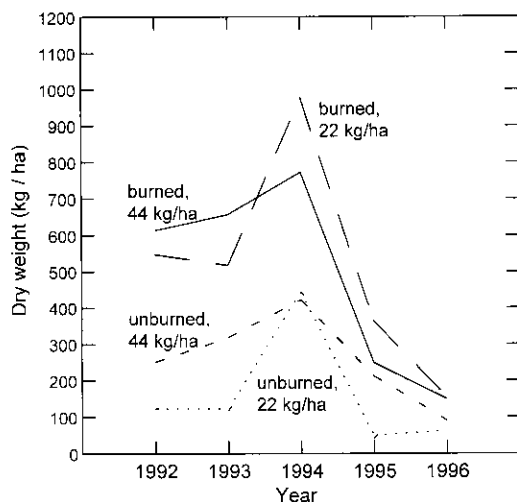


Figure 2. Mean dry weight production (kg/ha) of seeded orchardgrass and ryegrass from 1992 through 1996 in clearcuts with spring burning and seeding treatments.

markedly during the third year ( $P = 0.001$ ), then declined during the subsequent 2 seasons to negligible amounts ( $P < 0.001$ ). Doubling the rate of seeding from 22 kg/ha to 44 kg/ha had no overall effect on average annual production for the 4-year period, and annual differences between the two seeding rates were not different.

## Plant Communities

### Native Species

Richness of native species increased in all plots over time ( $P < 0.001$ ), but, increased more rapidly in burned than in unburned plots ( $P < 0.001$ ) (Figure 3a, b). Burned areas initially had fewer native species than unburned areas ( $P = 0.006$ ), but richness more than doubled in burned areas during the subsequent 5 yr, while increasing only about 50% in unburned areas. At the end of 5 yr, burned areas averaged about three more native species than unburned areas ( $P = 0.002$ ). Native species richness was unaffected by seeding treatments or by any interaction between seeding and burning.

Cover of native plants likewise increased with time after clearcutting ( $P < 0.001$ ) (Figure 3c, d), with both burned and unburned areas recovering at similar rates. After 5 yr, however, burned sites had 10% less native species cover than unburned

sites ( $P = 0.001$ ). Seeding treatments affected native species cover differently in burned and unburned areas ( $P = 0.063$ ). In burned areas, seeding (both levels combined) and fertilizing reduced native cover slightly ( $\sim 3.5\%$  over 5 yr) ( $P = 0.005$ ) compared to unseeded burned areas. In contrast, seeding and fertilizing unburned areas resulted in 12% higher native cover than in unseeded unburned areas ( $P < 0.001$ ). Positive responses of residual plants, primarily deer fern (*Blechnum spicant*) and residual or planted western hemlock seedlings, to seeding and fertilizing unburned areas appeared primarily responsible for the 12% treatment effect. However, another likely reason was that initial (1992) native plant richness and cover in those same unburned and seeded areas was nearly twice that of unburned unseeded areas ( $P = 0.001$ ).

### Exotic Species

More exotic species (about two species) were found in burned than unburned areas ( $P = 0.018$ ) (Figure 4a, b). Trends in exotic species richness also differed between burned and unburned areas ( $P = 0.054$ ). In burned areas, the number of exotic species increased during the first 3 yr, whereas, in unburned areas the number of exotic species remained relatively constant. Seeding had no effect on the number of exotic species, even though seed of five species of exotic grasses and legumes was introduced to seeded plots. There was no interactive effect of seeding and burning.

Burning increased exotic species cover by 3.5 times compared to unburned areas ( $P = 0.094$ ). Cover of exotic species was higher in seeded than unseeded areas ( $P = 0.038$ ) (Figure 4c, d), even though the number of exotic species was unaffected by the seeding treatment. Exotic species cover decreased in seeded areas after 1994 ( $P = 0.023$ ) as seeded grass cover dropped to low levels, while exotic species cover in unseeded areas increased slightly during that period ( $P < 0.001$ ). There was no interactive effect of burning and seeding on exotic species cover.

### Multi-Species Compositional Analysis

Species assemblages differed among treatment plots as a result of burning and seeding (intrasite  $r > 0.84$  with the first two ordination axes) (Table 3, Figure 5). Burning had the strongest influence on species composition (Axis 1, 60-66% of trace

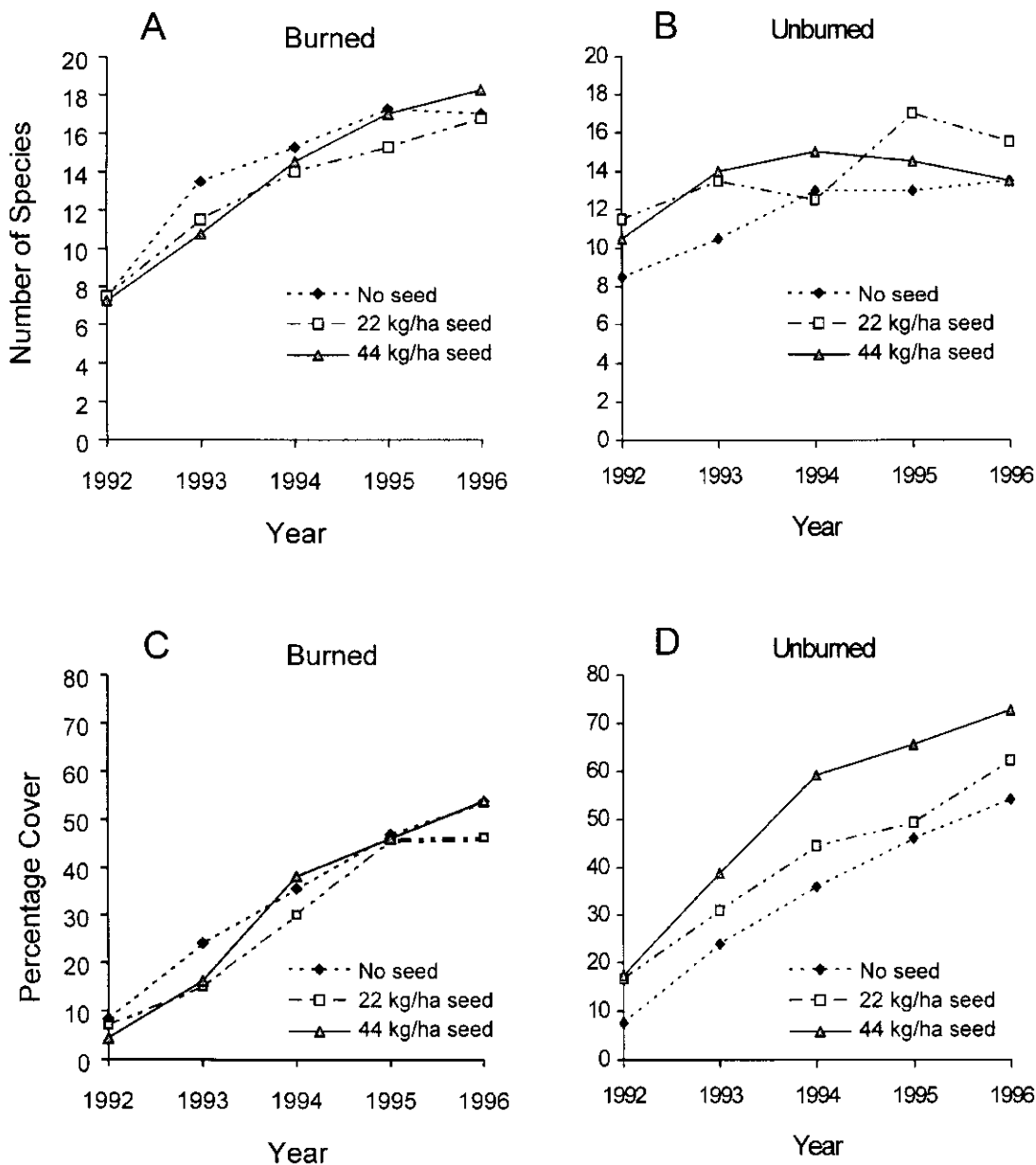


Figure 3. Species richness and canopy cover of native plant species from 1992 through 1996 in clearcuts with spring burning and seeding treatments.

variation across years). Seeding treatment had a weaker effect on species composition (Axis 2, 24-30% of trace variation across years).

Treatment effects weakened and species assemblages became more similar over time. Differences among plots were strongest in 1992 immediately after treatment, when treatments

accounted for 64% of the variation (i.e., the trace of ordination) in species composition among plots (Table 3). Plots became more similar over the subsequent 4 yr, as indicated by progressively smaller ordination traces and eigenvalues, and a shift of treatment centroids (group means) toward the origin of the ordination space (Figure 5). By

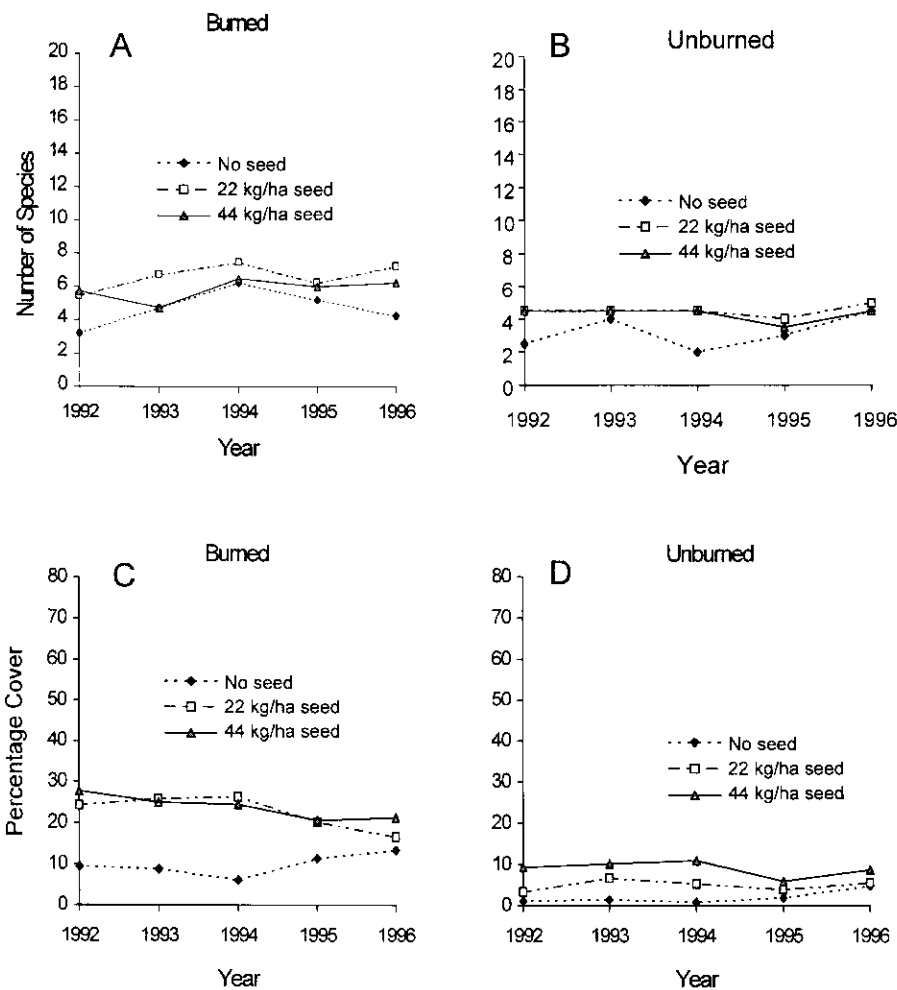


Figure 4. Species richness and canopy cover of exotic plant species from 1992 through 1996 in clearcuts with spring burning and seeding treatments.

TABLE 3. Canonical correlation analysis of the effects of spring burning and grass seeding rates on plant assemblages in clearcuts. All trace values and first eigenvalues (second eigenvalues were not tested) were significant ( $P \leq 0.10$ ) as determined by Monte Carlo permutation tests.

	1992 Axes		1994 Axes		1996 Axes	
	1	2	1	2	1	2
Trace <sup>a</sup>	0.638		0.378		0.212	
Eigenvalue	0.405	0.191	0.250	0.093	0.128	0.062
Percentage variance <sup>b</sup>	63.4	29.9	66.2	24.7	60.6	29.1
Intraset correlation <sup>c</sup>						
Burned	-0.940	0.333	-0.845	0.537	-0.954	0.403
Unburned	0.940	-0.333	0.845	-0.537	0.954	-0.403
No seed	0.265	0.857	0.555	0.845	0.308	0.852
22 kg/ha seed	-0.002	0.069	-0.238	-0.192	0.090	0.124
44 kg/ha seed	-0.202	-0.728	-0.230	-0.515	0.369	0.901

<sup>a</sup> Sum of four eigenvalues. Proportion of variation in species data accounted for by ordination.

<sup>b</sup> Percentage of total explained variance (trace) accounted for by ordination axes.

<sup>c</sup> Correlation coefficients of environmental variables with ordination axes.

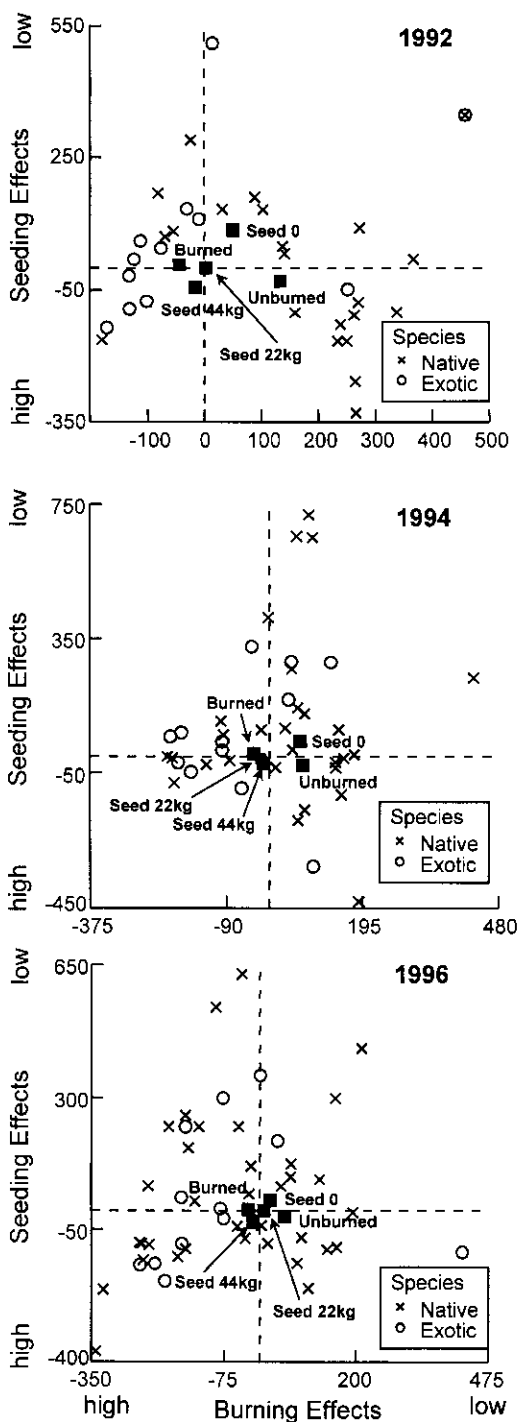


Figure 5. Biplots of plant species ordination scores and the centroids of treatment scores estimated by canonical correspondence analysis in clearcuts with spring burning and seeding treatments during 1992, 1994, and 1996.

the fifth year, only 21% of the variation (trace) in species composition among plots was associated with treatments. Exotic species continued to be most abundant in burned areas, whereas the distribution and abundance of native species became poorly associated with burning history, as indicated by a broad distribution of species along the primary burning gradient (Axis 1) (Figure 5).

#### Seedling Survival and Growth

An average of 25% of the planted western hemlock seedlings died over three growing seasons (1993-95) (Table 4). Nearly all (87-100%) mortality occurred during the first two growing seasons after planting. Burning and seeding treatments alone had no effects on seedling mortality, but their interaction affected seedling mortality in contrasting ways ( $P = 0.053$ ).

TABLE 4. Mean cumulative proportion of dead western hemlock seedlings planted in clearcuts with spring burning and seeding treatments. Analysis of variance tested results for 1995 only. Means with similar superscripts were not significantly different ( $P > 0.10$ ). The interaction between burning and seeding treatments was significant.

Treatment	Year	Seeding Rate			Mean
		No seed	22 kg/ha	44 kg/ha	
Burned	(n=131)	(n=142)	(n=110)	(n=383)	
	1993	0.184	0.286	0.289	
	1994	0.202	0.332	0.328	
	1995	0.202	0.332	0.328	0.287 <sup>a</sup>
Unburned	(n=57)	(n=39)	(n=75)	(n=171)	
	1993	0.328	0.202	0.123	
	1994	0.328	0.202	0.123	
	1995	0.328	0.202	0.123	0.218 <sup>a</sup>
Mean	(1995)	0.265 <sup>a</sup>	0.267 <sup>a</sup>	0.225 <sup>a</sup>	0.252

Seedling mortality in burned areas was 64% higher in seeded and fertilized plots than in unseeded plots ( $P < 0.01$ ) (Table 4). In contrast, seeding (combined effect of both rates) and fertilizing unburned areas decreased mortality by 31% compared to unseeded unburned areas ( $P < 0.001$ ).

Damage by wildlife, both ungulates and small mammals, appeared to be a minor factor in western hemlock seedling mortality. Only 3.8% of the 554 tagged seedlings were browsed, and only 1.4% of all western hemlock seedlings were both

browsed and dead at the end of the study. After 5 yr, cumulative browsing incidence was higher in unburned areas (5.4%) than in burned areas (3.4%) ( $P = 0.011$ ). Seeding and fertilization had no effect on incidence of browsing, and there was no interactive effect of burning and seeding on browsing.

Treatment effects on western redcedar seedling mortality were not tested because of an incomplete distribution of seedlings among treatment plots. However, simple inspection of the data regardless of treatments showed that western redcedar seedlings had slightly less mortality than did western hemlock, with about 21% mortality among the 327 tagged western redcedar seedlings. Western redcedar seedlings were more heavily browsed than western hemlock, with 74.3% incurring some browsing, primarily by mountain beaver (*Aplodontia rufa*) or snowshoe hare (*Lepus americana*). About 12% of all tagged western redcedar seedlings were both browsed and dead vs. 1.4% for tagged western hemlock.

Initial heights of planted western hemlock seedlings were not different in treatment plots, but after 3 yr seedlings were about 4% taller in burned than in unburned areas ( $P = 0.062$ ) (Table 5). Seeding treatments had no effect on final seedling height, nor was there an interactive effect of burning and seeding rate.

TABLE 5. Mean height (cm) of planted western hemlock seedlings after 3 growing seasons (1992–1995) in clearcuts with spring burning and seeding treatments. Means with different superscripts were significantly different ( $P \leq 0.10$ ). The interaction between burning and seeding treatments was not significant.

Burn treatment	Seeding rate			Mean
	No seed	22 kg/ha	44 kg/ha	
Burned	277	269	286	277 <sup>a</sup>
Unburned	273	283	250	267 <sup>b</sup>
Mean	276 <sup>a</sup>	273 <sup>a</sup>	273 <sup>a</sup>	274

## Discussion

### Forage Production

As hypothesized, burning enhanced the production of seeded forage grasses (Figure 2). Higher production in burned areas likely was a function of better germination conditions from reduction of duff and woody debris to ash (Klock et al. 1975,

Tiedemann and Klock 1976, Agee 1993), an immediate flush of available nutrients in burned sites from reduction of organic matter (Grier et al. 1989, Agee 1993), and less competition from native plants (Stewart et al. 1984, McDonald 1986, Owston et al. 1992) than in unburned sites. Unburned sites had initial post-treatment (1992) plant cover nearly double that of burned areas, and subsequent growth of residual native plants was rapid; hence, competition with residual vegetation likely limited forage production in unburned vs. burned areas. Bell et al. (2000) found that herbaceous plant cover was a good indicator of competitive relations between herbaceous and woody species.

Doubling the rate of seeding did not affect production (Figure 2). Consequently, managers could not compensate for poor conditions for grass germination or growth in unburned areas by increasing the seeding rate. Grass production in burned areas, primarily orchardgrass, was low compared to other studies. Peak production of about 1,000 kg/ha biomass in spring burned areas was reached during the third year after a relatively heavy seeding of 22 kg/ha. In contrast, Powell et al. (1994) measured 1,251 kg/ha production of orchardgrass from only 0.5 kg seed per ha applied to clearcut and fall-burned sites in drier lodgepole pine (*Pinus contorta*) and Engelmann spruce (*Picea engelmannii*) sites in interior British Columbia.

### Plant Communities

The hypothesis that spring burning reduces the richness and cover of native plants over a 5-yr period was rejected (Figure 3). The primary driver of successional change in native plants was the removal of the canopy by clearcutting and the subsequent growth of residual native species (Dyrness 1973, Alaback 1982, Stein 1995, Halpern and Spies 1995), primarily deer fern, salal, salmonberry (*Rubus spectabilis*), and western hemlock seedlings. Immediately after burning, native plant species richness and total cover were lower in burned plots than unburned plots. Within 5 yr, however, recovery of native species richness and cover differed little between unseeded burned and unburned stands. Species richness was initially lower in burned sites than unburned sites in the Oregon Cascades and Coast Range, but differed little between treatments after 4–7 yr (Dyrness 1973, Halpern 1988, Schoonmaker and McKee 1988,

and Stein 1995). Likewise, shrub cover was immediately lower in burnt than in unburnt stands in the central Oregon Cascades, but did not differ after 7 yr (Steen 1966). However, shrub and herb species compositions varied after 7 yr, depending on the species dominating the site after treatment.

Burning increased the richness and cover of invading exotic species as predicted (Figure 4). Dyrness (1973) and Halpern (1988) found similar differences in exotic species cover between lightly burned and unburned sites. They observed that differences among treatments were a function of the interaction between treatment effects and species life history and phenology. Similarly, in this study exotic species cover declined in burned areas as seeded grass cover declined, whereas in similarly burned, but unseeded, areas increasing exotic herb cover drove overall increases in exotic species cover.

The presence of seeded forage grasses had little apparent long-term effect on native plant communities (Figure 4). Grass biomass declined to negligible levels after 5 yr. The only effect of the seeding treatment on native plants occurred in unburned areas, where native species cover counter-intuitively was higher in seeded than in unseeded plots. Initially higher native species cover on unburned seeded vs. unseeded plots could explain much of that difference. Fertilization of unburned and seeded plots, as a component of the seeding treatment, also might explain the greater response of native species to seeding unburned plots. However, the response of common native species to fertilization varies by species and fertilizer composition (Prescott et al. 1993, Thomas et al. 1999, He and Barclay 2000). No conclusions about individual native species responses can be confidently made from this small sample.

#### Seedling Survival and Growth

Western hemlock seedlings grew taller (4%) in burned than unburned areas as predicted (Table 5). That result supports site preparation practices that eliminate competing vegetation to maximize western hemlock seedling survival and growth (Scott 1962, Ruth 1976). Seedling survival was affected by burning, but only if the site was seeded afterward (Table 4). On burned sites, seedling mortality was 64% higher with seeding than without seeding, most likely because of competition

with seeded grasses. Western hemlock is a shallow rooted species (Packee 1990) with slow seedling growth rates (Ruth 1976), and orchard grass is a strong competitor with conifer seedlings (Tiedemann and Klock 1976, Clark and McLean 1978, Powell et al. 1994) even at <10% foliar cover (Tiedemann and Klock 1976). Cover of seeded grasses in burned and seeded plots was 15% and 23% during the first two growing seasons after planting when most mortality occurred. On unburned sites, however, the opposite was true. Mortality was higher on unseeded sites, where fertilizer had not been applied as part of the seeding treatment, than on seeded and fertilized sites, possibly as a result of greater competition for nutrients with native species (Stewart et al. 1984, McDonald 1986, McDonald and Radosevich 1992). These complex relationships point to the need for controlled manipulative experiments to test for competitive interspecific interactions between species among treatments (e.g. Powell et al. 1994, Halpern et al. 1997).

#### Herbivory

Both small mammals (snowshoe hare and mountain beaver) and ungulates browsed western hemlock seedlings during the first year after planting. Burning affected herbivory, but seeding did not appear to reduce (Campbell and Evans 1975) or facilitate (Crouch 1969, Hines and Land 1974, Becker 1989) herbivory. Relatively high cover of residual vegetation in unburned vs. burned plots facilitated seedling herbivory by providing cover for small herbivores (Lawrence et al. 1961, Owston et al. 1992). In burned areas, herbivory was low because residual cover was poor and because seeding did not produce grass swards dense enough for small mammal cover. Cover apparently was insufficient to facilitate a population eruption of voles (*Microtus* spp.), which can effectively control woody vegetation through herbivory (Lawrence et al. 1961, Haufler and West 1986, Askham 1992).

In conclusion, many of the hypothesized treatment effects were supported either as main effects or interactions. Spring burning of clearcuts had relatively greater impacts than seeding and fertilizing on plant communities. High forage seeding and fertilizer application rates and their associated costs, the short productive period, poor production in unburned areas, and competition with conifer seedlings in burned areas should be weighed against the relatively short-term benefits

of forage seeding. Results of this study likely would apply to similarly treated western hemlock plant associations in coastal forests. However, managers should apply these results with caution and the understanding that substantial variation may occur in species responses to management treatments depending on site conditions (Scott 1962), species life histories, and variation in disturbance effects (Halpern 1988, Halpern et al. 1997). The complexity of the discovered treatment effects and the need to examine species competitive interactions more carefully both suggest the need for more extensively replicated and narrowly defined studies.

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