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Effects of forest harvesting, logging debris, and herbicides on the composition, diversity and assembly of a western Washington, USA plant community

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ABSTRACT

We examined plant community organization over the first five growing seasons after clearcut harvesting with retention of two levels of logging debris (light and heavy) and application of four vegetation control treatments (non-sprayed control, aminopyralid (A), triclopyr (T), and A + T). Our study site was 44 km northwest of Olympia, WA., USA, and before forest harvesting it was dominated by *Pseudotsuga menziesii*. We used a randomized split plot experimental design replicated in 6 blocks (each debris-treatment main plot had four herbicide-treatment split plots). We estimated percent canopy cover by species before forest harvesting and in post-harvest seasons 1–3 and 5 on 100 m^2 plots centered in each split plot. We analyzed species composition and diversity and report the response of 10 species groups and several major species to the treatments over five seasons. We used ANOVA to examine annual treatment effects on abundance of major species and species groups as well as ordination and graphical methods to examine succession. Abundance of ruderal species, especially exotics and graminoids, was lower but abundance of native woody shrubs and vines was higher in heavy debris than in light debris. The vines developed higher cover in heavy debris where they used debris as a scaffold to gain a competitive advantage over other species. Heavy debris controlled *Cytisus scoparius* better than the herbicide treatments. Triclopyr reduced woody dicot, vine and native herb covers, while aminopyralid reduced these groups and *Cytisus scoparius*, but aminopyralid had less effect on total canopy cover. The combination herbicide treatment reduced woody dicots, vines and *Cytisus scoparius*, and had the biggest impact on total canopy cover. By year 5 there was little difference in total canopy cover among the herbicide treatments; however, for some species, both debris and herbicide treatment effects were still apparent. The ordination indicated that, by the fifth season, the floristic characteristics distinguishing the debris treatments were still distinct, but not for the herbicide treatments. We conclude that heavy debris is a viable treatment alternative to prevent aggressive exotic species from competing with planted conifers and the native plant community on edaphically dry western Washington sites.

1. Introduction

Silvicultural techniques for forest plantation establishment are primarily designed to ensure crop tree success. However these techniques, including vegetation management with herbicides and logging debris removal, strongly affect all of the vegetation on the site. These techniques are often applied after forest harvesting causing considerable modification of both vegetation and microenvironment and leaving many species in a stressed condition. Plant stress caused by forest harvesting can be manifested in three ways: (1) physical breakage from falling or yarded trees, (2) burial under slash, and (3) physiological damage to shade adapted foliage now exposed to full sunlight and lower humidity (Halpern et al., 2005). While the results of these stressors are readily observable, they have not been studied under field conditions. The physiological mechanisms for breakage and burial seem obvious, but for foliar damage, it is presumably related to photoinhibition, which is only well-studied in laboratory settings (Long and Humphries, 1994, Powles, 1984). While field studies of stress physiology would be useful, managers still need generalized information about the combined effects of multiple management actions on vegetation to gain insight into alternative management methods.

Forest harvesting practices can be designed to contribute to a

Abbreviations: A, aminopyralid; T, triclopyr; A + T, combination of aminopyralid and triclopyr in a single treatment; C, non-sprayed control

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greater diversity of forest stand structures, wildlife habitats, and the maintenance of plant species (Halpern and Spies, 1995). Designing stand establishment techniques that consider all of the vegetation in terms of local and landscape biodiversity, wildlife habitat, soil productivity and aesthetics has become increasingly important, but the supporting plant community response information is often lacking.

Forest harvesting affects plant community composition (Peter and Harrington, 2009, Harrington et al., 2013), as well as soil processes (Powers et al., 2005, Nambiar, 1990, Vitousek and Matson, 1985), and concerns have been raised about its effects on species composition and biodiversity (Duffy and Meier, 1992, deMaynadier and Hunter Jr., 1995, Ito et al., 2006). Following forest harvesting, vegetation control is often used to increase soil water and nutrient availability to newly planted trees (Harrington and Tappeiner, 1991, Dinger and Rose, 2009), thereby improving their growth and survival (Fleming et al., 2006, Wagner and Robinson, 2006). Yet, the floral effects of vegetation control are difficult to generalize. Studies of plant diversity following herbicide applications have found reductions (Wilkins et al., 1993, Peter and Harrington, 2009), minimal change (Boateng et al., 2000, Freedman et al., 1993, Sullivan et al., 1998, Pitt et al., 2004, Hawkins et al., 2013) or increases in plant diversity (Lund-Hoie and Gronvold, 1987). This is not surprising considering the many different kinds of forest floras; the range of environmental conditions under which they grow; the presence of native and non-native floral components; the different kinds of herbicides and application methods, rates and timings; and the scale at which results are assessed. However, all of these studies and others (Wagner and Robinson, 2006, Peter and Harrington, 2012) note changes in species composition.

The amount of logging debris left on the site also affects vegetation after forest harvesting. Logging debris alters soil water and nutrient availability, soil temperature, and physical seedbed conditions (Powers et al., 1990). It is commonly left on site after harvest, but may also be removed to facilitate planting or for bioenergy production (Janowiak and Webster, 2010). Logging debris is never evenly distributed, so microsites are created and seed beds are modified depending on its depth and size distribution. Although a number of studies indicate minimal effects from removing logging debris on tree growth in soils of moderate to high fertility (Powers et al., 2005, Ponder et al., 2012, Scott et al., 2014), negative effects have been reported on soils of low fertility (O'Hehir and Nambiar, 2010) or where highly intensive practices were used (Egnell and Valinger, 2003; Smith et al., 2000). While some effort has been expended to examine effects on timber productivity, there has been little investigation into effects on early seral vegetation, although Harrington and Schoenholtz (2010) showed that logging debris removal could enhance exotic species invasion. Thiffault et al. (2011) suggests that, aside from the obvious effect of removing organic material and its associated nutrients, logging debris removal causes surficial soil disturbance with attendant effects on microclimate and vegetation. If research into the effects of logging debris removal and vegetation control on early seral vegetation is scant, information on the interactions of these practices when applied together shortly after forest harvesting is nearly non-existent.

In this paper we consider the effects of high or low levels of logging debris combined with single applications of two commonly used herbicides (aminopyralid and triclopyr) on the first five years of vegetation development following forest harvesting on a site of moderate forest productivity in the Pacific Northwest, USA. In this context, our study objective was to identify the effects of the various treatment combinations on community organization.

2. Methods

2.1. Study site

The Dry Bed Creek study site is located 44 km northwest of Olympia, WA and 2.6 km south southwest of Matlock, Washington on Green Diamond Resource Company land (47.215°N, 123.417°W). The site is flat, 37 m above sea level and receives 2381 mm of annual precipitation (PRISM Climate Group, 2017). The soil series is Grove gravelly sandy loam which is a deep somewhat excessively drained glacial outwash soil. This soil is classified as a sandy-skeletal, mixed, mesic Dystric Xerorthent (Soil Survey Staff, 2016). The dominant plant association is *Tsuga heterophylla* (Raf.) Sarg./*Gaultheria shallon* Pursh/*Xerophyllum tenax* (Pursh) Nutt. (Henderson et al., 1989). Prior to forest harvesting in 2011 the site had a mature *Pseudotsuga menziesii* (Mirb.) Franco *var. menziesii* overstory with a *Gaultheria shallon* and *Pteridium aquilinum* (L.) Kuhn dominated understory. The dominant trees ranged in age from 45 to 96 years. The stand was damaged in an ice storm in 1996 which was followed by a salvage harvest at that time. This opened canopy gaps which had only recently closed prior to the 2011 clearcut harvest associated with our study. *Cytisus scoparius* (L.) Link and many other ruderal species were introduced to the understory in these gaps.

2.2. Experimental design

The study was initiated in the summer of 2011 with the layout of forty-eight 100 m^2 circular vegetation measurement plots (5.64 m radius). Each vegetation plot was centered in a 21 m × 18 m herbicide treatment split plot. The overall experimental design is a randomized complete block with a split-plot treatment arrangement and six replications of eight treatments. The split plot design provides considerable power for testing treatment interactions. There were six blocks x two logging debris treatments x four herbicide treatments. Blocks were assigned according to distance from the main logging road, because preharvest vegetation inventories indicated that abundance of *Cytisus scoparius* decreased with distance from the road. Clusters of the four herbicide treatment split plots were overlaid on each of the debris main plots.

Clearcut harvesting occurred in December 2011 with chainsaws. The logging debris treatments were carried out immediately after log removal while the needle debris was fresh and green. An excavator with a clamshell bucket was used to create two levels of debris. The light debris treatment was achieved with whole-tree harvesting, but occasionally excess debris from limb breakage was moved from these areas to areas intended for the heavy debris treatment. For the heavy debris treatment, trees were limbed and bucked on the plot and only the boles were removed. In February 2012, logging debris depth averaged 32 and 17 cm for heavy and light debris treatments respectively, based on 21 systematically located points per split plot. Line-transect estimates of logging debris mass (Brown, 1974) conducted on each split plot in August 2012 indicated mean values of 18.9 and 9.0 Mg ha^{-1} for the heavy and light debris treatments, respectively.

Even though the yarder lifted each bole to remove it from the plot, observation of the yarding process indicated that some branches still dragged on the ground when whole trees (with limbs) were removed causing soil disturbance and residual vegetation damage that did not occur when limbed boles were removed. Although we did not quantify soil scarification, it appeared to be minimal compared to operational shovel logging. However, this small difference in scarification should be considered to be part of the heavy and light debris treatments.

Clusters of four herbicide-treatment split plots (two pairs separated by a 10 m wide harvest machine trail) were included in each logging debris-treatment main plot. Each herbicide treatment split plot was randomly assigned one of four herbicide treatments: (1) an experimental control with no herbicide; (2) 0.5 L ha^{-1} of aminopyralid (Milestone®) in water with 0.25% Syl-Tac® surfactant; (3) 2.9 L ha^{-1} of triclopyr ester (Forestry Garlon® XRT) in water with SuperSpread® methylated seed oil at 2.5% concentration; (4) the combination of the two herbicides. The herbicides were applied uniformly to assigned plots on August 13, 2012 (after the 2012 vegetation measurement) with backpack sprayers at a spray volume of 94 L ha^{-1}. Triclopyr and aminopyralid were chosen because they are commonly used in the forest

industry and because they have different modes of action. Triclopyr is absorbed through foliage, but aminopyralid is principally a soil active herbicide affecting dicotyledonous germinants as they emerge. Each of the herbicides has little or no activity on established monocotyledonous plants.

The first vegetation measurement was made in June 2011 prior to forest harvesting. The overstory was removed during November and December 2011. The second vegetation assessment was made in June 2012 prior to the herbicide applications. In February 2013, *Pseudotsuga menziesii* seedlings (1 + 1 stocktype) from a local seed source were hand-planted with shovels on a 3 × 3-m grid over the entire installation. Vegetation assessments were also carried out in June of 2013, 2014 and 2016. We visually estimated percentage of canopy cover for each vascular species in the overstory and understory (Henderson et al., 1989). Using this method, a single species' cover is constrained to a maximum of 100%, but the sum of the species' covers can exceed 100% due to crown overlap. After clearcutting, all *Pseudotsuga menziesii* cover (planted and naturally regenerated) was combined, but planted trees accounted for nearly all of the cover. Density of new *Cytisus scoparius* seedlings was estimated for each split plot as the mean value from four 0.1 m^2 quadrats located at cardinal directions 3 m from plot center. *Cytisus scoparius* seedling density was measured in July of 2012, 2013, 2014, and 2015.

2.3. Ordination analysis

We used NMS ordination (PC-Ord version 5) (McCune and Mefford, 2006) to evaluate community floristic change in relation to the treatments and over time. The ordination input was canopy cover for each species on each plot in each year of the study (240 plot samples and 125 species). We used the default settings for PC-Ord (Sorenson distance measure, 6 axes, 500 iterations, random starting coordinates, reduction in dimensionality of 1 at each cycle, and a step length of 0.20). Autopilot and thoroughness were specified. This resulted in a 3-dimensional solution with final stress = 13.35526. We present the ordination results as the calculated average value for each treatment group of plots (the treatment centroid) in each year along the first two axes of the ordination. For a measure of the relative amount of floristic change from year to year for each treatment, we analyzed the change in the position of the treatment centroids from year to year with a two factor analysis of variance (ANOVA) (logging debris and herbicide treatments and their interaction). To aid in the interpretation of the axes we present the five strongest tau correlations of the most strongly correlated groups and species.

2.4. Analysis of variance

Individual species were combined into ten functional groups for analysis purposes (Table 1). We analyzed the summed canopy cover of these aggregated functional species groups, as well as the individual covers of the most common species. Criteria for defining a group included: growth habit (tree, shrub, herb), taxonomic group (monocot, dicot, conifer or fern), and nativity (native or exotic). Individual species were also selected for ANOVA if their average cover after forest harvesting (2012–2016) equaled or exceeded 0.5%, resulting in selection of 17 species.

We summed the species covers for each functional group on each plot for each year. Functional group and major species cover data were subjected to repeated-measures ANOVA in Proc Mixed (SAS Institute, Inc., 2013) to test the significance of the fixed factors, logging debris treatment, herbicide treatment, measurement year, and their interactions, after adjusting for random effects of blocks. We also included the pre-harvest canopy cover as a covariate whenever its significance was $P \leq 0.1$. For species cover variables (bounded by a maximum of 100%), an angular transformation was applied to proportionate values of cover to homogenize their residual variances (Sokal and Rohlf, 1981) prior to their use in the ANOVA. However, cover values reported in text, tables and figures are non-transformed data. Residuals for each response variable were plotted against predicted values to check for non-homogenous variance. If a year-by-treatment interaction was detected, slicing was used in SAS to identify individual years in which differences existed among treatments. When treatment differences were detected, multiple comparisons of adjusted means were conducted with Bonferroni probabilities to control the Type I error rate (Quinn and Keough, 2002). We consider P values less than or equal to 0.05 to be statistically significant.

Because herbicide treatments were applied after the 2012 vegetation assessments, we analyzed the 2012 canopy cover data for debris effects with single factor ANOVA similar to what has just been described, but without the repeated measures component. Where significant differences were detected we used the Tukey HSD test for mean separations. We analyzed richness, evenness, Shannon, and Simpson diversity indexes using the same approach and included pre-harvest (2011) values of each variable as covariates in the ANOVAs when $P \leq 0.1$. Shannon's index is more sensitive to richness and rare species, while Simpson's index is more sensitive to dominance and evenness.

3. Results

3.1. Pre-harvest effects and initial post-harvest responses

The 2011 forest stand had a closed *Pseudotsuga menziesii* canopy with occasional canopy gaps and a uniform understory dominated by *Gaultheria shallon*, *Polystichum munitum* (Kaulf.) C. Presl, *Pteridium aquilinum* and *Rubus ursinus* Cham. & Schltdl. The vines (dominated by *Rubus ursinus*), are not strong climbers, but rather ramble over other vegetation and debris. They do not twine and they have no attachment mechanism other than thorns. *Rubus ursinus*, rambled over *Gaultheria shallon* and other plants to gain height and competitive advantage in the pre-harvest stand.

Falling trees during clearcutting and branch dragging during yarding crushed or broke most understory plants. In the first year after

Table 1
Group names used in other figures, their sensitivity to the herbicides used in the study, and several characteristics of each group.

Group names	Number of species in each group	Their sensitivity	Group characteristics
Exotic shrubs	3	Yes	Dicots; early seral, intolerant to moderate shade tolerance; mostly *Cytisus scoparius*
Evergreen trees	5	Yes	Mid to late seral native coniferous dominants; mostly *Pseudotsuga menziesii*
Woody dicots	22	Yes	Early to late seral woody stemmed native shrubs; intolerant to highly shade tolerant
Vines	2	Yes	Dicots; early to late seral native species, moderately shade tolerant; mostly *Rubus ursinus*
Exotic graminoids	10	No	Monocots; early seral heliophytes
Exotic herbs	19	Yes	Dicots; early seral heliophytes
Native herbs	29	Yes	Dicots; early to late seral, moderate to high shade tolerance
Monocot herbs	6	No	Early to late seral native species
Native graminoids	18	No	Monocots; early to mid-seral
Ferns	7	No	Early to late seral, moderate to high shade tolerant native species

Table 2
F-statistic probabilities from the 2012 and the 2013–2016 analyses of variance of cover for plant species and species groups, diversity indexes, or ordination axes to detect potential effects of debris (D), herbicides (H), year (Y) and their interactions. Shown in bold face type are the relevant significant debris or herbicide probabilities for each response variable (year and covariate probabilities are not bolded).

	Source of variation								
	2012	2013–2016							
	D	D	H	DxH	Y	DxY	HxY	DxHxY	Cov[1]
		(Numerator/denominator degrees of freedom)							
Response variable[2]	(1/5)	(1/5)	(3/30)	(3/30)	(2/80)	(2/80)	(6/80)	(6/80)	(1/80)
	Probability > F								
Total	0.381	0.118	0.001	**0.045**	0.001	**0.040**	**0.001**	0.088	NS
Evergreen trees	0.111	0.313	0.339	0.306	0.671	0.291	0.395	0.306	NS
Woody dicots	0.102	0.049	**0.001**	0.066	0.001	**0.007**	0.082	0.488	0.008
Monocot herbs	0.088	0.234	0.196	0.258	0.001	0.102	0.293	0.290	0.001
Native graminoids	**0.039**	0.144	0.093	0.120	0.088	0.388	0.518	0.767	0.001
Native herbs	0.931	0.132	**0.017**	0.848	0.001	0.623	0.093	0.411	0.042
Vines	0.105	**0.014**	0.001	0.911	0.001	0.540	**0.001**	0.481	0.063
Ferns	0.148	0.517	0.425	**0.049**	0.001	**0.011**	0.691	0.662	0.001
Exotic graminoids	**0.016**	**0.004**	0.157	0.068	0.068	0.536	0.192	0.371	0.021
Exotic herbs	**0.007**	0.016	0.012	0.702	0.001	0.673	0.035	**0.019**	NS
Exotic shrubs	0.344	0.080	**0.038**	0.338	0.001	**0.001**	0.371	0.294	NS
Agrostis capillaris	0.330	0.076	0.003	**0.042**	0.001	0.586	0.495	0.5140	0.028
Amelanchier alnifolia	0.389	0.146	**0.001**	0.339	0.001	0.927	0.471	0.241	0.001
Anthoxanthum odoratum	0.391	0.007	0.407	**0.006**	0.381	0.569	0.215	0.272	0.001
Bromus vulgaris	0.428	0.662	0.672	0.291	0.001	**0.032**	0.706	0.549	0.001
Crepis capillaris	0.354	0.054	0.058	0.250	0.001	**0.001**	0.094	0.270	NS
Cytisus scoparius	0.379	0.013	0.052	0.593	0.001	**0.001**	0.368	0.302	0.001
Elymus glaucus	0.415	0.089	0.263	0.367	0.001	0.401	0.054	0.916	0.001
Gaultheria shallon	0.653	0.559	0.001	0.212	0.001	**0.002**	**0.001**	0.207	0.008
Hypochaeris radicata	0.329	0.061	0.568	0.902	0.001	**0.001**	0.322	0.373	NS
Leucanthemum vulgare	0.334	0.344	0.017	0.259	0.001	0.748	0.001	**0.012**	0.020
Linnaea borealis	0.493	0.123	0.320	0.542	0.001	**0.009**	0.093	0.546	0.001
Polystichum munitum	0.416	**0.050**	0.082	0.101	0.001	0.320	**0.042**	0.422	0.001
Pseudotsuga menziesii	0.111	0.860	0.153	0.667	0.001	0.459	0.064	0.293	NS
Pteridium aquilinum	0.505	0.126	0.778	0.372	0.001	**0.013**	0.728	0.395	0.001
Rubus ursinus	0.332	**0.013**	0.001	0.961	0.001	0.713	**0.001**	0.515	0.075
Symphoricarpos hesperius	0.366	0.018	0.001	**0.020**	0.001	**0.028**	0.270	0.332	0.001
Veronica chamaedrys	0.810	0.054	0.058	0.250	0.001	**0.001**	0.094	0.270	0.045
Richness	**0.015**	0.086	0.208	0.711	0.001	**0.006**	0.001	0.198	0.016
Evenness Index	0.150	0.177	0.803	**0.026**	0.001	**0.035**	0.881	0.378	NS
Shannon Index	0.340	0.113	0.474	**0.048**	0.001	**0.014**	0.922	0.203	NS
Simpson Index	0.182	0.088	0.640	0.001	0.001	0.081	0.742	**0.044**	NS
Ordination Axis 1	0.478	0.678	**0.012**	0.583	0.001	**0.001**	0.144	0.541	0.001
Ordination Axis 2	0.232	0.181	0.001	0.271	0.001	0.171	**0.001**	0.557	0.035
Ordination Axis 3	0.076	**0.003**	**0.001**	0.092	0.018	0.284	0.156	0.244	NS

[1] The covariate, pre-treatment (2011) values for a given response variable, was included in the analysis of variance when $P \leq 0.1$; NS = not significant ($P > 0.1$).

clearcutting, logging debris was the tallest remaining structure, and in the heavy debris treatment, it was the densest and most continuous cover. *Rubus ursinus* rapidly utilized this new debris scaffolding to grow above other sprouting or recovering species.

Variation in the pre-harvest understory left an imprint on the developing post-harvest plant community as shown by the frequency of significant covariates used in our 2013–2016 ANOVA (Table 2). The covariate was significant in 7 of the 10 functional groups and in 14 of 17 major plant species. The groups and species with significant covariates were more-or-less evenly distributed over the future treatment groups. However, by random chance, the plots designated to receive the heavy debris/aminopyralid treatment had the most *Gaultheria shallon* in 2011 (89%) and the lowest mean cover of *Symphoricarpos hesperius* G.N. Jones (8%). Due to high *Gaultheria shallon* dominance, these plots were low in exotic shrub, exotic herb, exotic graminoid, native graminoid, monocot herb and native herb covers compared to other plot groups resulting in the lowest diversity (Table 3). The distinction of being low in diversity, highest in *Gaultheria shallon* (19%) and lowest in *Symphoricarpos hesperius* (22%) continued through 2016 as did lower than average covers for all of the other previously mentioned groups.

3.2. Ordination

Our ordination was solved in 3 dimensions, although Fig. 1 shows only the two axes along which the most annual change occurred. The movement of the annual centroids trace a generalized transition from forest to the harvested stand, and the subsequent succession upon which we imposed treatments.

The groups of plots assigned to receive the treatments were floristically most homogenous in 2011 prior to forest harvesting (Fig. 1). The largest floristic change for all plots was due to the timber harvest as measured by the distance between the mean ordination centroids of 2011 and 2012 (Fig. 1, Table 4) and is aligned with axis 2. The strongest species correlations with axis 2 are negative, and with forest species and groups (Table 4). Resprouting and growth of forest residual species is reflected in negative movement along axis 2 through 2014. After 2012 centroid movement increased along axes 1 and 3 reflecting increasing dominance of ruderal species (Fig. 1). Most strong correlations with axis 1 were positive and associated with ruderal species and groups (Table 4). Axis 3 (not shown), was similar to axis 1 in that it reflected recovery processes. In its case, negative values represent ruderal species

Table 3
Diversity indexes for the debris x herbicide treatment combinations and the main effects means. Underlined data show where significant differences exist between debris treatments according to the 2012 and 2013–2016 ANOVAs (Table 2). Superscripts indicate significant differences for interactions in the 2013–2016 ANOVA (Table 2). C = non-sprayed control, A = aminopyralid, T = triclopyr, A + T = combination of aminopyralid and triclopyr. Heavy = heavy debris and Light = light debris.

	Heavy				Light				Main effect treatment means					
	C	A	T	A + T	C	A	T	A + T	Heavy	Light	C	A	T	A + T
Richness														
2011	22.0	19.5	23.5	22.8	23.0	18.8	20.5	22.8	22.0	21.3	22.5	19.2	22.0	22.8
2012	21.3	18.3	20.8	22.8	25.7	22.8	24.2	23.3	20.8	24.0	23.5	20.6	22.5	23.1
2013	30.2	26.5	30.7	29.5	34.7	28.2	30.7	32.5	29.2	31.5	32.4^{a}	27.3^{b}	30.7^{ab}	31.0^{ab}
2014	31.2	30.8	32.3	33.2	34.2	30.3	33.7	36.8	31.9	33.8	32.7	30.6	33.0	35.0
2016	31.8	30.2	30.3	29.5	29.5	31.0	28.8	30.0	30.5	29.8	30.7	30.6	29.6	29.8
Evenness														
2011	0.59	0.56	0.56	0.59	0.58	0.62	0.62	0.59	0.58	0.60	0.58	0.59	0.59	0.59
2012	0.56	0.55	0.61	0.61	0.55	0.55	0.50	0.55	0.58	0.54	0.55	0.55	0.55	0.58
2013	0.49	0.47	0.54	0.56	0.53	0.56	0.51	0.47	0.52	0.52	0.51	0.51	0.53	0.52
2014	0.50	0.47	0.52	0.54	0.56	0.57	0.56	0.51	0.51	0.55	0.53	0.52	0.54	0.53
2016	0.56	0.52	0.55	0.59	0.57	0.57	0.58	0.57	0.56	0.57	0.56	0.55	0.56	0.58
Shannon														
2011	1.80	1.65	1.77	1.85	1.81	1.80	1.84	1.83	1.77	1.82	1.81	1.73	1.81	1.84
2012	1.70	1.61	1.81	1.92	1.78	1.71	1.59	1.75	1.76	1.71	1.74	1.66	1.70	1.83
2013	1.68	1.52	1.85	1.89	1.86	1.85	1.74	1.63	1.73	1.77	1.77	1.69	1.79	1.76
2014	1.73	1.60	1.80	1.88	1.98	1.92	1.97	1.85	1.75	1.93	1.85	1.76	1.88	1.86
2016	1.93	1.78	1.88	1.98	1.92	1.96	1.94	1.95	1.89	1.94	1.92	1.87	1.91	1.96
Simpson														
2011	0.79	0.75	0.77	0.79	0.78	0.79	0.80	0.79	0.78	0.79	0.78	0.77	0.78	0.79
2012	0.71	0.68	0.77	0.78	0.71	0.69	0.64	0.72	0.73	0.69	0.71	0.69	0.70	0.75
2013	0.70^{ab}	0.65^{a}	0.77^{ab}	0.78^{b}	0.78^{b}	0.79^{b}	0.73^{ab}	0.66^{a}	0.72	0.74	0.74	0.72	0.75	0.72
2014	0.73^{ab}	0.67^{a}	0.76^{ab}	0.78^{ab}	0.80^{b}	0.80^{b}	0.79^{ab}	0.74^{ab}	0.73	0.78	0.77	0.74	0.78	0.76
2016	0.77	0.74	0.77	0.81	0.78	0.81	0.80	0.78	0.77	0.79	0.77	0.77	0.79	0.80

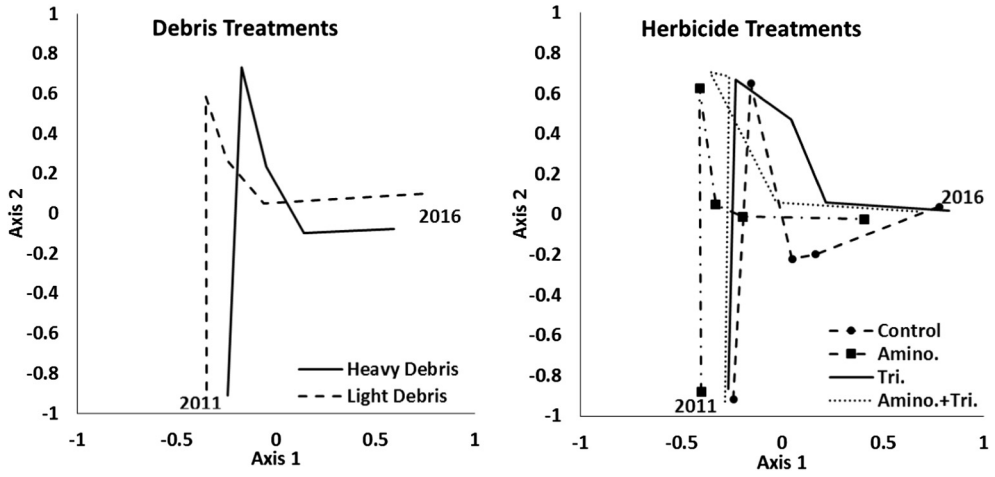

Fig. 1. Successional diagrams based on nonmetric multidimensional scaling of all species, plots and years. Shown are the mean centroids for the plots in each of the treatments by year. Each endpoint or bend represents the centroid for a sample year, thus each segment represents the relative amount (length) or direction (angle) of floristic change between years. These plots represented a single forest community in 2011 prior to forest harvesting. The first angle in each line represents the 2012 sample (after forest harvesting and debris treatments). The second angle represents the 2013 sample (first sampling after herbicide treatments).

and groups, and positive values represent recovering residual species and groups (Table 4).

Floristic change between 2011 and 2012 was significantly greater in the heavy than in the light debris treatment suggesting the importance of burial effects of the debris treatments (Fig. 2a). From 2014 to 2016 the pattern changed, with significantly more floristic change in the light debris plots than in the heavy debris plots.

Between 2012 and 2013 there was more floristic change in the non-sprayed control than in the herbicide treated plots (Fig. 2b). However, between 2013 and 2014 the most floristic change occurred in the combination treatment and was least in the non-sprayed control. Floristic change continued from 2014 to 2016 but with little difference among the herbicide treatments (Fig. 2b).

The treatments were reflected in plot groupings along the ordination axes. On axis 1, plots were significantly grouped by herbicide treatment and by a debris x year interaction (Table 2). Triclopyr plot locations were significantly more positive than those of the aminopyralid or combination treatments. This difference was established in 2013, and was maintained through 2016 (Fig. 1). Thus, greater ruderal dominance established in the first year post treatment was maintained through 2016. The debris x year interaction is visible as a more positive (on axis 1) light debris centroid location than for heavy debris in 2016, suggesting that ruderal groups are more important in the light debris community. Axis 2 had a significant herbicide x year interaction. In 2013, the combination and triclopyr plot locations were significantly more positive than the non-sprayed control and aminopyralid locations suggesting greater recovery of residual species on non-sprayed control and aminopyralid plots. Similarly, the herbicide main effect on axis 3 indicated that combination and triclopyr plot locations were significantly more negative than the non-sprayed control and aminopyralid locations (Table 2). This also suggests greater recovery of residual species and groups on non-sprayed control and aminopyralid plots. The significant debris main effect on axis 3 (Table 2) suggests greater recovery of residual species on heavy debris plots.

Table 4
The five strongest group and species correlations (tau) for each NMS ordination axis.

Axis 1		Axis 2		Axis 3	
Exotic herbs	0.421	Native herbs	−0.183	Woody dicots	0.324
Exotic shrubs	0.364	Ferns	−0.336	Exotic shrubs	−0.197
Exotic graminoids	0.283	Woody dicots	−0.449	Native graminoids	−0.388
Native graminoids	0.227	Evergreen trees	−0.476	Exotic herbs	−0.471
Ferns	−0.382	Vines	−0.553	Exotic graminoids	−0.592
Symphoricarpos hesperius	0.648	*Senecio sylvaticus L.*	0.386	*Gaultheria shallon*	0.550
Leucanthemum vulgare	0.503	*Gaultheria shallon*	−0.316	*Rosa gymnocarpa*	0.393
Crepis capillaris	0.427	*Frangula purshiana*	−0.340	*Hypochaeris radicata*	−0.418
Elymus glaucus	0.404	*Pseudotsuga menziesii*	−0.473	*Leucanthemum vulgare*	−0.443
Cytisus scoparius	0.371	*Rubus ursinus*	−0.562	*Anthoxanthum odoratum*	−0.565

April-September of 2015 and 2016 only had 40 to 75% of the precipitation observed during the same months of 2012, 2013, and 2014 (Harrington et al., 2018). Planted *Pseudotsuga menziesii* cover decreased from 0.5% in 2013 and 2014 to 0.2% in 2016. Although we did not quantify it separately from the cover estimates, we noted visible dieback and mortality of many residual species including *Gaultheria shallon*, *Linnaea borealis L.*, *Rosa gymnocarpa* Nutt., *Polystichum munitum*, *Pteridium aquilinum*, *Rubus ursinus* and *Xerophyllum tenax* among others. Some ruderal species also became prematurely desiccated, but either survived underground (graminoids and some perennial herbs) or regrew from seed (many annual, biennial and perennial herbs). We saw no mortality of *Cytisus scoparius* but some plants had signs of crown dieback and reduced flowering.

3.3. Species diversity

Species richness was lowest in 2011 (21.63), peaked in 2014 (32.81) and declined slightly in 2016 (30.15). Evenness was highest in 2011 (0.59), lowest in 2013 (0.52), then increased to 2016 (0.56). Shannon's index started at 1.80 in 2011, fell to 1.73 in 2012, and rose to 1.92 in 2016. Simpson's index began and ended at 0.78 (2011 and 2016) and was lowest in 2012 at 0.71.

The diversity indexes were more sensitive to debris than to herbicide treatments, but only richness had a significant herbicide x year interaction (Table 2). The non-sprayed control had the highest richness in 2013 (32.4), but it was only significantly different from the aminopyralid (27.3) treatment.

Richness was the only diversity index to differ significantly between debris treatments in 2012 (light 24.0, heavy 20.8), a pattern that continued in 2013 (light 31.5, heavy 29.2, herbicide x year interaction, Table 2). Evenness and Shannon's index were higher in light than in heavy debris in 2014 (evenness: 0.6 light, 0.5 heavy; Shannon's: 1.9 light, 1.8 heavy). Simpson's index followed a similar annual pattern as Shannon's index, but with no significant differences (Table 2). An inverse pattern emerged for richness compared to the other diversity indexes for the years 2012–2016. Richness was higher in light debris than in heavy debris for 2012–2014 and lower in 2016. The opposite was true for evenness, Shannon's and Simpson's indexes: (lower in light debris than in heavy debris in 2012 and the opposite in 2013–2016).

There were similar debris x herbicide interactions in the evenness and Shannon's indexes (Table 2). These indexes increased for heavy debris in the following treatment sequence: aminopyralid, triclopyr, non-sprayed control, combination treatment, while the indexes for light debris were flat or decreased over the same sequence. The indexes for aminopyralid, and to a lesser extent triclopyr, were higher with light than with heavy debris, but in combination, the indexes were higher with heavy debris than with light debris. Thus, the combination of these herbicides did not magnify their individual effects, but rather changed the effect altogether.

There was a significant debris x herbicide x year interaction in Simpson's Index with significant differences in 2013 and 2014. Thereafter, the indexes for all treatments tended to converge. In 2013, the light debris/non-sprayed control, light debris/aminopyralid, and heavy debris/combination treatments were higher than the heavy debris/aminopyralid or light debris/combination treatments. Although the values were already converging in 2014, indexes for the light debris/non-sprayed control and light debris/aminopyralid treatments remained higher than the heavy debris/aminopyralid treatment. Thus, there are similarities in the response of all three indexes. These interactions may be an artifact originating in pre-treatment differences in the 2011 forest stand. Plots that later received the heavy debris/aminopyralid treatments already had low diversity indexes in 2011 and these differences were maintained, more-or-less in proportion throughout the experiment.

3.4. Vegetation cover responses to logging debris

Total cover (sum of all species covers) averaged across all treatments in 2011, 2012, 2013, 2014 and 2016 was 293, 47, 108, 134 and 147% respectively. In 2012 there was no difference in total cover by debris treatment, but differences emerged in 2013. In 2013, there was significantly less total cover in heavy debris (98%) than in light debris (119%) (Table 2), but thereafter total covers converged resulting in a significant debris x year interaction. There was also a debris x herbicide interaction. The response of total cover to triclopyr and the combination treatment was similar in heavy and light debris, but there was more

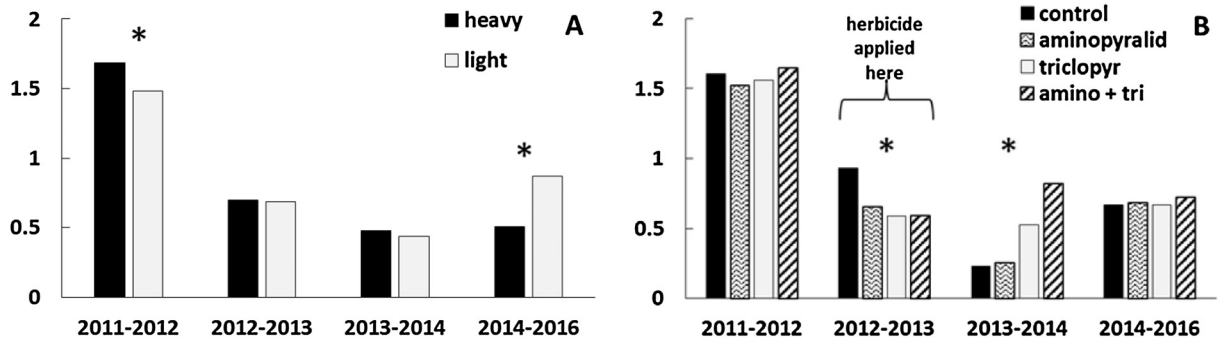

Fig. 2. Mean magnitude of floristic change according to an NMS ordination by: (A) logging debris treatment and (B) herbicide treatment. This figure refers to the ordination depicted in Fig. 1. In this case the y axis is the difference in position of the ordination centroid between the two years indicated (length of the vector between the two years on the x and y axis in Fig. 1). The interval of 2011–2012 includes the forest harvesting and debris treatments. Herbicide treatments were applied in the 2012–2013 interval. An asterisk denotes intervals in which significant treatment differences occurred (ANOVA, $P \leq 0.05$).

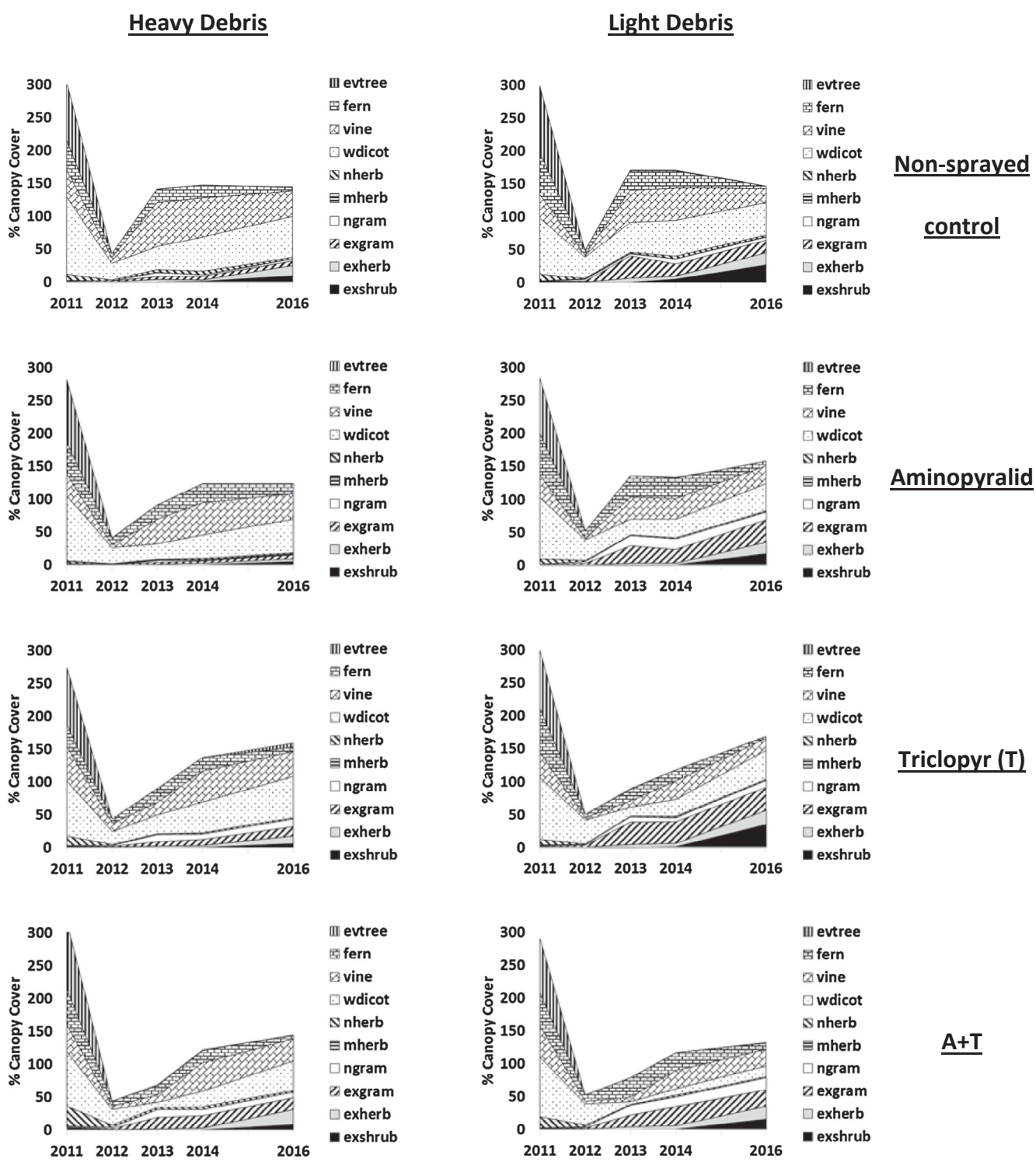

Fig. 3. Sums of functional group canopy cover by year. The data are treatment means (n = 6) of the sums of canopy cover for all species in each of 10 functional groups. No data were collected in 2015.

cover in light than in heavy debris with the aminopyralid or non-sprayed control treatments.

Among the species groups and abundant species, there were more significant responses to debris treatments than to herbicide treatments (Table 2). Exotic graminoids averaged 0.5% in the 2011 forest stand, increased in 2012 to 1% (heavy debris) and 2% (light debris), and maintained significantly higher cover in light debris (26–28%) than in heavy debris (8–11%) from 2013 through 2016 (Fig. 3). Two exotic graminoids with significant debris x herbicide interactions were *Agrostis capillaris* L. and *Anthoxanthum odoratum* L. (Table 2, Fig. 4). *Agrostis capillaris* was released by the combination herbicide treatment, but only in the light debris treatment. *Anthoxanthum odoratum* was suppressed by heavy debris, but triclopyr was more effective at releasing this species than the combination treatment.

Vine cover prior to forest harvesting (2011) was 40% and decreased to 7.6% (high debris) and 3.7% (low debris) in 2012. From 2013 to 2016 vine cover remained significantly higher in heavy debris (34–50%) than in light debris (23–34%) (Fig. 3). *Rubus ursinus* had

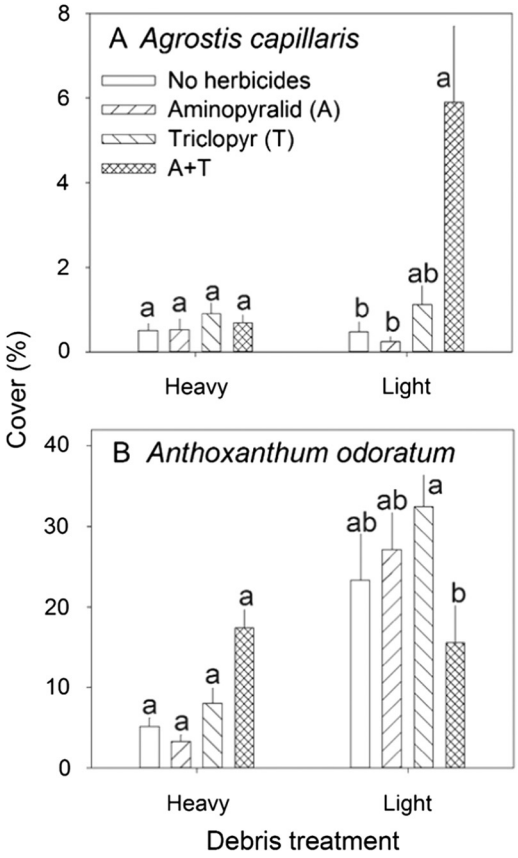

Fig. 4. Cover responses (± standard errors) for the debris x herbicide interaction from the analysis of variance for: (A) *Agrostis capillaris* and (B) *Anthoxanthum odoratum*. Letters indicate differences among herbicides within a given debris treatment. See Table 2 for *F*-statistic probabilities.

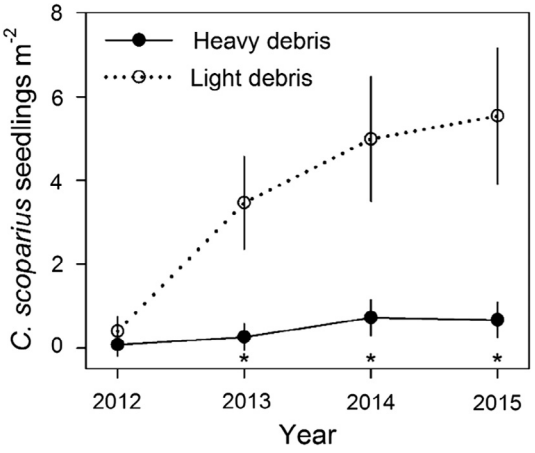

Fig. 6. Mean density of new *Cytisus scoparius* seedlings by logging debris treatment and year.

significantly more cover over the 2013–2016 interval in heavy than in light debris (Table 2, Fig. 5). Cover for *Rubus ursinus* peaked in 2014 and then decreased in both debris treatments due to the 2015 and 2016 droughts.

Covers of exotic shrubs and woody dicots diverged between the light and heavy debris treatments with differences becoming significant in 2016. In 2016, exotic shrub cover averaged 7% in heavy and 24% in light debris while woody dicot cover averaged 55% in heavy debris and 38% in light debris. The dominant exotic shrub, *Cytisus scoparius*, had significantly more cover in light than in heavy debris in 2016 (Fig. 5). While we were not able to discern differences in *Cytisus scoparius* cover in earlier years, seedling counts showed that debris effects on density of new *Cytisus scoparius* seedlings were already present from 2013 onward, although effects of the herbicide treatments were not detected (data not shown; $P \geq 0.30$) (Fig. 6). For one of the dominant woody dicots, *Symphoricarpos hesperius*, cover diverged between the light and heavy debris treatments with differences becoming significant in 2014 and

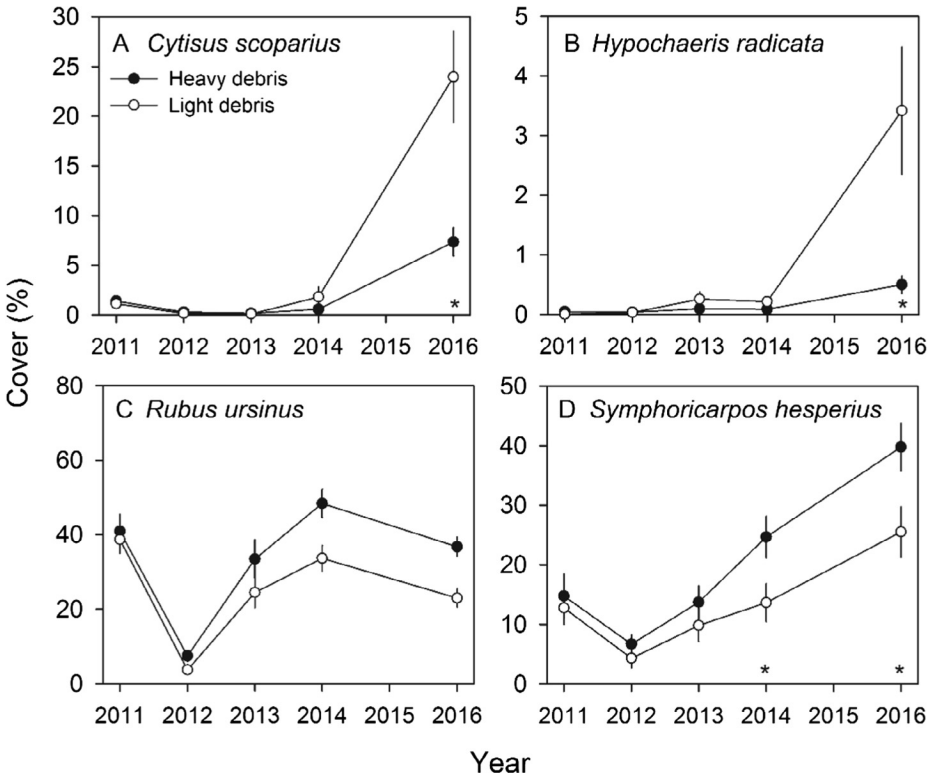

Fig. 5. Species covers (± standard errors) for all years and significant results from the 2013–2016 analysis of variance. Debris by year interactions were significant for (a) *Cytisus scoparius*, (b) *Hypochaeris radicata*, and d) *Symphoricarpos hesperius*. *Rubus ursinus* (c) had a significant debris treatment main effect. Asterisks (*) indicate years in which species' cover differed between debris treatments. See Table 2 for *F*-statistic probabilities.

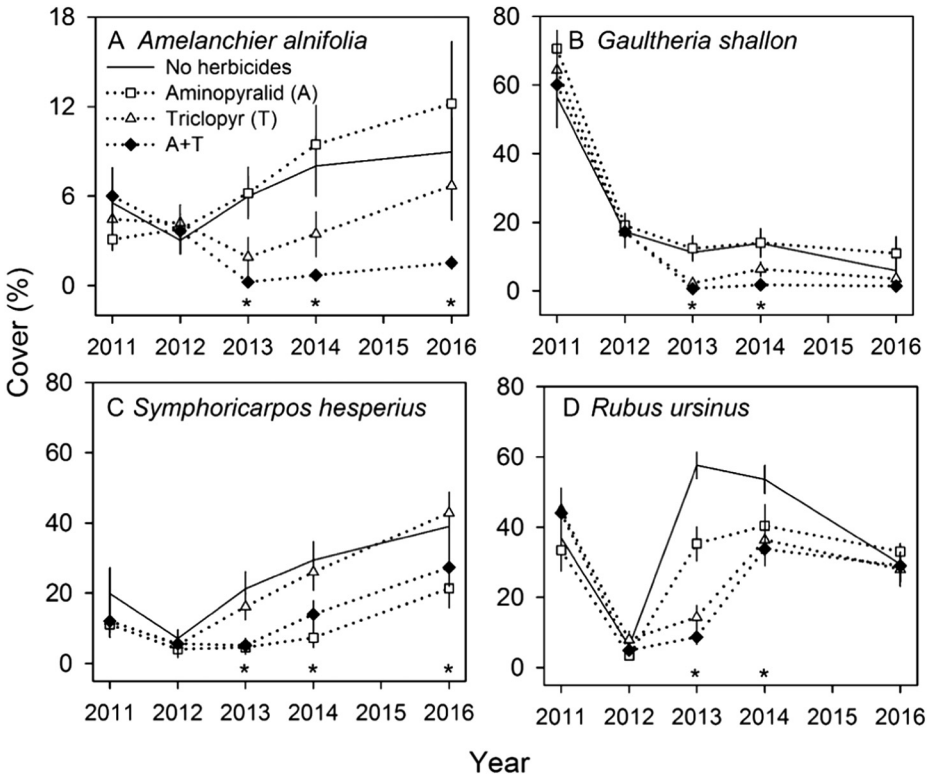

Fig. 7. Cover responses (± standard errors) for the herbicide x year interaction from the analysis of variance for: (a) *Amelanchier alnifolia*, (b) *Gaultheria shallon*, (c) *Symphoricarpos hesperius*, and (d) *Rubus ursinus*. Asterisks (*) indicate years in which species' cover differed among herbicide treatments. See text for a description of treatment differences and Table 2 for *F*-statistic probabilities.

2016 (Fig. 5).

Exotic herbs, like exotic graminoids, had significantly more cover in light debris (0.5%) than in heavy debris (0.2%) in 2012, but there was a debris x herbicide x year interaction in the 2013–2016 ANOVA caused by divergence of treatment responses from 2014 to 2016. Exotic herbs tended to have less cover in heavy debris, but both the difference and the cover increased after 2014. The increase in cover after 2014 was likely due to the droughts. However, the amount of divergence was influenced by the herbicide treatment. In 2016, the heavy debris/aminopyralid treatment had significantly lower cover (4.4%) than other treatments except the heavy debris/triclopyr (11.4%) or heavy debris/non-sprayed control treatments (15.6%). These latter two treatments did not differ from any other treatment.

The fern group had a significant debris x year interaction, but the largest difference between treatments, which occurred in 2013 (19.3% heavy debris; 27.4% light debris), was not significant. Following 2013 the difference between treatments began to converge, and following 2014 the cover in both treatments decreased due to drought. There was also a difficult to interpret debris x herbicide interaction. In all treatments except triclopyr, fern cover was higher in light debris than in heavy debris in 2013. However, this was reversed in the triclopyr treatment, which appears to be a reflection of pre-treatment (2011) cover rather than a debris or herbicide effect.

Four species groups (evergreen trees, monocot herbs, native graminoids and native herbs) had no debris response in the 2013–2016 ANOVA. However, native graminoids had significantly more cover in light debris (2%) than in heavy debris (1%) in 2012.

3.5. Vegetation cover responses to herbicides

After forest harvesting, total canopy cover accumulated fastest in the non-sprayed control and light debris/aminopyralid treatments between 2012 and 2013. Total cover then plateaued in the non-sprayed control, but it was significantly lower in the herbicide treatments in 2013 and 2014 (Table 2, Fig. 3). Accumulation was slower and spread over 2012 to 2014 in the triclopyr, combination, and heavy debris/aminopyralid treatments (Fig. 3). Total cover in the herbicide treatments converged with that in the non-sprayed control by 2016 resulting in a significant herbicide x year interaction.

Without herbicide treatments, the cover of most species groups rapidly increased in the year after overstory removal before leveling off (Fig. 3). With herbicide treatments, dicotyledonous groups increased more slowly than without herbicides and in some cases decreased between 2012 and 2013 (e.g., woody dicots, Fig. 3). The combination treatment, in particular, retarded the development of canopy cover for most dicot groups through 2013, but graminoid cover increased. After 2013, recovery of dicot groups is indicated by a sharp increase in cover.

Exotic shrubs had significant herbicide main effects, and the herbicide x year interaction was not significant (Table 2). However, in 2016 the response was diverging, with cover averaging 18–20% in the triclopyr and non-sprayed control treatments and 11–12% in the aminopyralid and combination treatments (Fig. 3).

Native herb cover (all dicots) declined in the year after the herbicide treatments (2013), increased in 2014 and then decreased through 2016, presumably from drought. The non-sprayed control had the highest average cover at 3.5%, which was significantly greater than in the triclopyr and aminopyralid treatments (both 1.5%), but not the combination treatment (2.5%). While this species group had little cover it was the most diverse (28 species).

The *Gaultheria-shallon*-dominated woody dicot group was greatly reduced by the herbicide combination treatment by 2013, but although it then increased steadily it had significantly lower cover than all other treatments (Table 2, Fig. 3). Over the same time period woody dicot cover in the aminopyralid treatment was significantly lower than in the non-sprayed control, but not lower than the triclopyr treatment. The triclopyr treatment was not different from the non-sprayed control. The combination treatment and to a lesser extent the triclopyr treatment reduced cover of *Amelanchier alnifolia* (Nutt.) Nutt. ex M. Roem. in 2013, 2014 and 2016 (Fig. 7). *Gaultheria shallon*, a shade tolerant species that dominated the pre-harvest forest stand, responded similarly to *Amelanchier alnifolia*, but with a more muted response due to its slow recovery and the impact of the drought (Fig. 7). However,

Symphoricarpos hesperius was reduced by the aminopyralid and to a lesser extent the combination treatment in the same years (Fig. 7).

The vine species group had a significant herbicide x year interaction due to convergence in cover among herbicide treatments after 2013. However in 2013, in the non-sprayed control, it had significantly higher cover than any other group (58%) followed by aminopyralid (35%), which was also different from all other groups. It did not differ between triclopyr (14%) and the combination treatment (9%). Overall, it increased to 2014, then leveled or decreased due to drought. Because most of the vine group cover consisted of *Rubus ursinus*, this species largely mirrored the vine group response (Fig. 7).

There were no significant responses to herbicides by exotic or native graminoids, evergreen trees, or monocot herbs.

4. Discussion

The concept of directional succession towards a site determined potential vegetation is well established in ecology (Daubenmire, 1968, Westman, 1978, Sutherland, 1981, Halpern and Franklin, 1987, Cook, 1996, Pickett et al., 2009). We assume that similar vegetation to the pre-harvest community will reassemble regardless of treatment, although by different pathways. The question dealt with here is how those pathways originate under different early seral management.

Compositional contrast between pre- and post-disturbance communities is determined by the nature and severity of the disturbance. Moderate disturbances may increase diversity, but if a single disturbance or the sum of successive disturbances is large enough, diversity will decrease (Connell, 1978). Increased diversity is found when site preparation techniques permit entry of early seral species without eliminating previously abundant late seral species (Swindel et al., 1984), but loss of diversity is found when high rates or repeated applications of herbicides have been used (Wilkins et al., 1993, Peter and Harrington, 2009, Peter and Harrington, 2012). Minimal loss is generally observed from herbicide applications when the treated community was already assembled after overstory removal (Freedman et al., 1993, Sullivan et al., 1998, Boateng et al., 2000, Pitt et al., 2004, Hawkins et al., 2013). Lund-Hoie and Gronvold (1987) found diversity increased in response to glyphosate application in young Norway spruce plantations, which they attribute to a change from a relatively homogeneous pre-spray perennial plant community to one with a higher proportion of annuals after application. Thus, results cannot be generalized without a firm understanding of the floral composition and contrast between the pre- and post-treatment communities. In our case, the pre-harvest forest canopy had re-closed relatively recently following a 1996 ice storm and salvage operation, meaning that the forest flora already contained an early seral component that might not have been present in a similar, but undisturbed stand. This effectively decreased the floral contrast with the post-harvest community. Also, the treatments were applied before adjustment to the overstory removal, so generalization must consider cumulative effects. Thus, although forest harvesting induced a clear plant community phase change (Li, 2002), floral contrast with the post-harvest community was moderate and species flux was less than what would occur if a well-developed second growth or old-growth community were similarly treated, but greater than if an assembled early seral community had been treated.

Forest management often involves multiple disturbances making it difficult to separate the effects of each activity. In this experiment, overstory removal with its associated soil disturbance was the first and arguably the most important disturbance. Halpern and Franklin (1987) found that soil disturbance early in succession profoundly affected the magnitude and direction of vegetation change. The sudden change in light and humidity, together with burial by debris and physical damage to understory plants and the forest floor stressed residual plants and provided resources for ruderal invaders. Thus, it is not surprising that most of the species gained were ruderal, often exotic, species that in some cases later became dominant. Most species that were lost were minor native forest understory species. It is noteworthy that the greatest rate of recruitment for new seedlings of *Cytisus scoparius* occurred in the second year after forest harvesting (2013), likely because of time needed to activate the seedbank.

4.1. Community floristic trends

Ordinations can clarify complex floristic trends (Fig. 1). The strong divergence from the 2011 forest community composition in 2012 was due to the nearly complete loss of *Pseudotsuga menziesii* as well as heavy harvest-associated understory damage and release of ruderal species. Burial of vegetation in the heavy debris treatment, but not the light debris treatment accounts for most of the difference in distance moved between the debris treatments. Then, in 2013, there was a partial convergence with the 2011 centroids in proportion to recovery of residual forest understory species, which was greater in the heavy debris treatment, and in the absence of herbicides. Thus, debris burial did not hinder and possibly encouraged recovery of forest understory species. Recovery was delayed a year or two in the triclopyr and combination treatments which killed or set back recovery of native shrubs and herbs. Over time, the ordination centroids moved increasingly to the right on axis 1, driven by the appearance and in some cases dominance of ruderal species. The triclopyr treatment moved more in this direction due to greater ruderal dominance encouraged by herbicide reduction of native residual species cover. However, positive movement on axis 1 was dominant between 2014 and 2016 in all groups due to drought, which reduced native fern and vine cover, opening space for exotic herbs and *Cytisus scoparius*. Although, drought was the driver, vegetation responded differently depending on treatment: herbicide effects faded in response to drought, but debris effects did not, suggesting that mulching effects of debris (e.g., soil water conservation) lessened the impact of the drought on native residual species.

The 2011 stand was dominated by *Pseudotsuga menziesii* in the overstory (87% cover) and woody dicots, ferns, and vines in the understory (Fig. 3). Forest harvesting greatly reduced the cover of these groups, largely by mechanically killing their tops. While the coniferous trees cannot resprout, many plants from the other groups along with many herbaceous species did resprout. With the exception of *Symphoricarpos hesperius*, the woody dicot group was slow to regrow, but regrowth of *Rubus ursinus* (the principal species of the vine group) and *Pteridium aquilinum* was comparatively rapid. Resprouting and growth of these residuals greatly accelerated the recovery of the forest community, reflecting the importance of initial floristics to this succession (Egler, 1954). While our study only lasted for 5 years, others found herbicide treatment effects on understory vegetation to be ephemeral (Maguire et al., 2009), lasting perhaps no more than 10 years (Wagner and Robinson, 2006). Our herbicide findings seem in line with these studies. Our ordination implies that debris treatment effects were more long lasting than the herbicide treatment effects. Similar debris mediated responses of vegetation were also found at a nearby site where *Cytisus scoparius* grew more abundantly in light than in heavy debris causing greater mortality to *Pseudotsuga menziesii* seedlings (Harrington and Schoenholtz, 2010). Large reductions of native species have been reported following *Cytisus scoparius* invasion (Wearne and Morgan, 2004) so the successional pathway back to native forest vegetation may be considerably longer in the light debris treatment, particularly if it results in greater *Pseudotsuga menziesii* mortality. The ability of forest shrubs and trees to overtop the ruderals when wetter seasons return will eventually renew dominance by native forest species. But these observations illuminate some of the reasons this may require many years, and how differences in early composition could pre-dispose a community toward enduring alternate pathways.

4.2. Logging debris effects

Whole-tree harvesting causes more soil disturbance and increases

light availability at the forest floor favoring ruderal species, but stem-only harvesting leaves debris that shades the ground, moderates soil temperatures, preserves soil moisture through mulching (Powers, 2002, Roberts et al., 2005, Trottier-Picard, 2014), and intercepts herbicides decreasing their effectiveness (Banks and Robinson, 1982, Reddy et al., 1995, Peter and Harrington, 2009). In our study, shade from heavy debris, still green in the early part of the first growing season, limited first season ruderal establishment but it favored recovery of the mostly shade tolerant forest flora. Residual forest species sprouting from well-established roots grew under or through the debris. By 2016, the mean woody dicot cover in heavy debris plots was 56% while that of the light debris plots was 38%. At the same time, covers of exotic graminoids and *Cytisus scoparius* were 11% and 7% in the heavy debris treatment, respectively, and 28% and 24% in the light debris treatment. Thus, heavy debris combined with less soil disturbance than in the light debris treatment enhanced forest species survival and recovery, and it discouraged ruderal species invasion, resulting in decreased diversity and a shortened forest community assembly pathway.

A notable residual forest species was the vine, *Rubus ursinus*, which rapidly spread through and over the top of the debris, producing lasting ground level shade after the needles on the debris had fallen, further suppressing heliophytic plants germinating under the debris. Rapid, dense spreading by this species in Pacific Northwest openings was also observed by Nesmith et al. (2006). The herbicides (especially triclopyr) reduced *Rubus ursinus* cover most in areas of light debris, probably due to herbicide interception by the heavy debris. Isensee et al. (1998) concluded that herbicide performance may be affected by the type and condition of residues on the soil surface at the time of application and that the amount of herbicide reaching its target may depend on interactions between the type and amount of crop residue and the amount of rainfall after the application. Without the debris, *Rubus ursinus* was not only exposed to more herbicide which effectively killed much of it, but was forced to compete at ground level with exotic ruderal monocots such as *Anthoxanthum odoratum* that were not affected by the herbicides and would not support the weight of the vine.

4.3. Herbicide effects

Residual shrubs frequently lost their tops during forest harvesting and as of 2012 were only just beginning to resprout. The combination of large underground root and rhizome systems with a small aboveground surface area resulted in canopy interception of a relatively low dose of herbicide that was not always lethal. Thus, many residual species were damaged, but survived the herbicide application to resprout in 2014. For these plants, the herbicide effect was limited to the loss of about one year's growth. However, fast growing ruderals, especially graminoids, were able to exploit this temporary suppression, and became more prominent in 2013 (Fig. 3).

The herbicide treatments were applied in late summer, after most residuals had ceased growth and many herbs were desiccated and dormant. Some results may have differed if the herbicide treatments had been applied earlier in the season during active growth, and before seed production, especially for triclopyr. Although triclopyr affects existing *Cytisus scoparius* plants, our treatment was ineffective because at the time of application most of this species existed as un-germinated seed. Aminopyralid controlled *Cytisus scoparius* because of its soil activity on germinating seedlings (Harrington, 2014).

By 2014 summed canopy covers exceeded 100% in all treatments suggesting an emerging role for resource competition, in particular, competition for light, as a successional driver. Changes in the resources that plants compete for often drive floristic change (Huston and De Angeles, 1994). Resource competition actually emerged as the major driver of succession in the second or third years after forest harvesting in the absence of herbicides, and perhaps in the aminopyralid treatment as well, but in the third or fourth growing season in the triclopyr and combination treatments.

However, following 2014, there was renewed successional divergence relative to the 2011 forest community (Fig. 1) precipitated by severe summer drought in 2015 and 2016. It is difficult to say if drought had anything to do with the emergence of *Cytisus scoparius* as a dominant species during this time, but it can be said that *Cytisus scoparius* tolerated the drought with limited visible effects. However, drought killed many native shrubs, ferns and *Pseudotsuga menziesii* seedlings (Harrington et al., 2018), facilitating ruderal and exotic species invasion, especially on the light debris plots (Fig. 1). During this time herbicide effects largely vanished, but debris effects remained, which reflects the inhibiting effect heavy debris had on graminoids and *Cytisus scoparius* and the greater level of soil disturbance in the light debris treatment.

4.4. Diversity

There were few differences in diversity between any of the herbicide treatments and the non-sprayed control. The reason for this appears to be a balanced shift from forest species to ruderal species. Of the four diversity indexes, only richness showed a difference and only in 2013. Others have also found compositional changes (often toward ruderal species), but little change in diversity (Freedman et al., 1993, Sullivan et al., 1998, Hawkins et al., 2013, Stein 1995).

There is at least some suggestion that triclopyr increased diversity, but the difference was slight and the effect of debris was greater. Richness differed the most between debris treatments in 2013, when more ruderal species were added than forest species were lost in the light debris treatment. The diversity indexes tended to be higher in light than in heavy debris from 2013 to 2016 (significantly so in 2014 for evenness and Shannon's index). This pattern results from the initial loss of native forest species due to the disturbance of forest harvesting followed by the recruitment of native and exotic ruderal species, especially in areas with light debris. Unequal distribution of debris created a mosaic of microsites with different resource potentials, so evenness tended to be lower in heavy debris. All of the indexes appeared to be converging by 2016 suggesting that the effects of these treatments on diversity are short-lived.

5. Conclusions

Treatment effects on diversity indexes were subtle, however both debris and herbicides strongly affected species composition. The debris treatment suppressed *Cytisus scoparius* and other exotic species, suggesting its use for preserving native vegetation and preventing mortality of planted trees. Different species responded in different ways to the herbicides. Aminopyralid suppressed germination of *Cytisus scoparius* (Harrington, 2014) and both herbicides reduced *Rubus ursinus*, but triclopyr was more effective than aminopyralid. *Amelanchier alnifolia* and *Gaultheria shallon* were suppressed by triclopyr, but not by aminopyralid and the reverse was true for *Symphoricarpos hesperius*. The sensitivity of mature *Symphoricarpos hesperius* to aminopyralid was unexpected. Thus, the risk of collateral damage when narrowly targeting a species such as *Cytisus scoparius* with this herbicide is greater than expected. Herbicides applied over heavy logging debris were less effective, which we attribute to spray interception by the debris, and possibly also to better conditions for recovery due to less stressful soil moisture conditions. However, since the logging debris suppressed *Cytisus scoparius* and other exotic species while ameliorating drought conditions for planted trees and native species, there is less need for vegetation control.

Although climatically moist, the Dry Bed Creek site is edaphically prone to summer drought. Under these circumstances light debris and herbicide treatments, and their combination, lengthen the successional pathway back to native forest composition by favoring competitive exotic species. Heavy debris provided an early advantage for the relatively shade tolerant native species, setting the community on a more

direct path to the original forest composition by effectively suppressing *Cytisus scoparius* and other competitive exotic species. It also encouraged the regrowth of planted *Pseudotsuga menziesii*, both by reducing competition with exotic species and by ameliorating drought (Harrington et al., 2018). We believe debris retention after forest harvesting to be a viable, cost saving alternative to herbicide use on these kinds of sites. As things now stand, it appears that *Cytisus scoparius* is poised for heavy dominance of the light debris treatment, causing delays in successional convergence because of increased mortality of planted *Pseudotsuga menziesii* seedlings (Harrington et al., 2018). Other studies (Halpern and Franklin, 1987) suggest that convergence with the original forest composition is likely, but our study suggests that this will occur through different pathways and in different amounts of time depending on the specific treatment combinations.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.foreco.2018.01.045.

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