Multi-scale Analysis of Fire Effects in Alpine Treeline Ecotones

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Abstract

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Although the direct effects of climate change have been studied though observational and experimental methods in alpine treeline ecotones (ATEs), indirect effects due to shifts in disturbance regimes have received less attention, despite evidence that the frequency and extent of large disturbances are increasing in many other ecosystems.

At a regional scale, I analyzed wildfires occurring over a 29-year period (1984-2012) in ATEs in eight mountainous ecoregions of the Pacific Northwest and Northern Rocky Mountains. I focused on two components of the ATE: (1) subalpine parkland, which extends from closed subalpine forest through a fine-scale mosaic of forests and non-forest, and (2) alpine vegetation, which includes meadow, shrubland, and alpine tundra. I expected that subalpine parkland and alpine vegetation would burn less, proportionally, than the entire ecoregion. In four of eight ecoregions—three in Rocky Mountains and one in the Cascades—the proportion of subalpine

parkland burned was comparable or greater than the proportion of the entire ecoregion that burned. In alpine ecosystems little of the area (<7%) burned during the 29-year study period.

At a local scale, I examined variability in fire severity and changes in plant structure, using data from >500 plots within four alpine treeline ecotones sites in the Cascade Range and Northern Rocky Mountains, which had burned 18-27 years prior. I assessed the likelihood of different pre-fire canopy-cover structural classes—closed forest (>40% tree cover), open forest (10%-40%), parkland (<10%), and unforested areas (alpine, meadow, and Krummholz)—to burn and to change to a different structural class after fire. I also evaluated changes in forest structure—specifically the abundance of live trees within five diameter at breast height (DBH) classes—using non-metric multidimensional scaling (NMDS) to visualize differences and Permutational Multivariate Analysis of Variance (PERMANOVA) to test statistically for differences from pre-fire to post-fire, and between unburned and three higher-severity class. Non-forested areas were less likely to burn and fire increased the proportion of non-forested area. The effects of the fire on forest structure were mixed: previously forested stands had a greater probability of retaining forest cover than they had of becoming non-forested. Greater fire severity decreased the abundance of larger, relative to smaller, overstory trees; the latter suffered greater mortality. Of the four common high-elevation tree species observed in burned plots, *Abies lasiocarpa* had the highest rates of mortality (60%), *Larix lyallii* had the lowest rate (11%), with intermediate levels in *Pinus albicaulis* (52%) and *Picea engelmannii* (37%).

Strong, significant correlations between the overall annual area burned across all vegetation types, and the area burned in subalpine parkland and alpine vegetation (ρ = 0.89 and ρ = 0.88, respectively) indicate that fire may become more prevalent in both subalpine parkland and alpine vegetation if the overall area burned increases due to climate change. Within burned ATEs, fire effects are moderate, and highly heterogeneous. The combined effect of climate change and fire may cause ATEs to expand upward and trees to infill previously snow-dominated sites, while simultaneously increasing fine- and course-scale heterogeneity within the ecotone, due to fire-cause mortality.

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Chapter 1: Area Burned in Alpine Treeline Ecotones (ATE) in the Pacific Northwest and Northern Rocky Mountains (1984-2012)

Abstract

The direct effects of climate change on alpine treeline ecotones (ATEs)—the transition zone between subalpine forest and non-forested alpine vegetation—have been studied extensively using observational and experimental approaches. Climate-induced changes in disturbance regimes have received less attention. Increases in the frequency and extent of large disturbances have occurred in many ecosystems. To determine if recent climate-driven increases in area burned are impacting ATEs, I analyzed wildfires occurring over a 29-year period (1984-2012) in ATEs in eight mountainous ecoregions of the Pacific Northwest and Northern Rocky Mountains. I focused on two components of the ATE: (1) subalpine parkland, which extends from closed subalpine forest through a fine-scale mosaic of forests and non-forest, and (2) alpine vegetation, which includes meadow, shrubland, and alpine tundra. I expected that subalpine parkland and alpine vegetation would burn less, proportionally, than the entire ecoregion. Across the Pacific Northwest and Northern Rocky Mountains 55,137 ha of subalpine parkland burned and 27,510 ha of alpine vegetation burned during the 29-year study period. However, the proportions burned were highly variable across regions, ranging from <1% to 22%. In four of eight ecoregions—three in Rocky Mountains and one in the Cascades—the proportion of subalpine parkland burned was comparable or greater than the proportion of the entire ecoregion that burned. In alpine ecosystems estimates of fire rotations were long (>400 years) and little of the area (<7%) burned during the 29-year study period, in all but one ecoregion. Area burned in both subalpine parkland and alpine vegetation was correlated with the total area burned in each ecoregion, suggesting that similar climatic drivers or the spread or fire from neighboring vegetation types strongly influences burning in the ATE. Future changes in total area burned are likely to affect the ATE.

Introduction

Across western North America, the area burned and frequency of large wildfires declined in the middle of the 20th century, but has increased since the 1970s. Annual variation in fire frequency and area burned are influenced, in part, by climate in forest ecosystems throughout western North America (Heyerdahl et al. 2008, Littell et al. 2009, Mori 2011, Abatzoglou and Kolden 2013). The Pacific Northwest and Northern Rocky Mountains, the focal regions of this study, have experienced a significant increase in mean annual temperature, modest increases in precipitation, reductions in snowpack, earlier snowmelt, and a longer freeze-free season (Mote et al. 2005, Johnstone and Mantua 2014, Jolly et al. 2015), all of which contribute to a longer fire season and increased area burned (Littell et al. 2009, Abatzoglou and Kolden 2013). These changes have occurred since 1900 (increasingly since 1980) and are due to both climate teleconnections (*i.e.*, El Niño–Southern Oscillation and Pacific North American pattern) and anthropogenic influences (Abatzoglou et al. 2014, but see Johnstone and Mantua 2014). Further increases in area burned are expected due to anthropogenic climate change (Flannigan et al. 2009, Littell et al. 2010), which will interact with ongoing changes in the behaviour and severity of fires caused by past timber harvest, grazing, and fire suppression (Hessburg et al. 2015).

Despite increasing evidence that fire regimes are responding rapidly to changes in climate, the extent of this response is not well understood. I focus on changes in fire regimes at the highest elevations of subalpine forests and in adjacent non-forest vegetation. This transitional zone, the alpine treeline ecotone (ATE) (Malanson et al. 2007), extends from the upper edge of continuous subalpine forest, through subalpine forests, to treeless alpine tundra (including Krummholz and tree islands). Here, I divide the ATE into two main vegetation types: *subalpine parkland*, which includes the upper boundaries of closed forest and adjacent fine-scale (<1 ha) mosaics of forest and non-forest vegetation (Rochefort et al. 1994), and *alpine vegetation*, the non-forested (>1 ha) areas of meadow, shrub field, and alpine tundra.

My focus on the ATE is motivated by observations that (1) this ecotone is expected to shift upward in direct response to climate change, and may already be doing so in some regions (Harsch et al. 2009); and (2) fire regimes in subalpine forests may be more responsive to changes in climate than those of other forest types. Many studies have assessed the influence of climate warming on the alpine treeline ecotone (Harsch et al. 2009), but little research has focused on how climate-driven changes in fire regimes may be modifying this influence.

Recent changes in the fire regimes of high-elevation forests (Westerling et al. 2006, Reilly 2014, Cansler and McKenzie 2014, Harvey 2015, Zhao et al. 2015), in which pre-settlement fire return intervals exceed the period of fire suppression, should reflect changes in climate (Littell et al. 2009, Mallek et al. 2013). In addition subalpine forests may experience larger increases in area burned than other forest types: when they are dry enough to burn, fire spread is aided by their characteristically greater continuity of fuel (Turner and Romme 1994, Bessie and Johnson 1995, Cansler and McKenzie 2014).

Dendrochronological and paleoecological studies indicate that subalpine parkland does burn, although these studies have not consistently differentiated closed subalpine forests from parkland. In the northern Rocky Mountains prior to Euro-American settlement, fire rotations in subalpine forests and parklands ranged from 175-350 years, with slightly shorter rotations in Montana and Idaho (~150-250) than east of the Continental Divide in Wyoming and Colorado (~250-350) (Baker 2009). Fire rotations in *Abies lasiocarpa* parkland in the Cascade Range (100-275 years) are similar to those in Montana and Idaho (Fahnestock 1976, Franklin et al. 1988, Agee et al. 1990). Others forest types, such as *Pinus albicaulis-A. lasiocarpa*, may have burned more frequently (Tomback et al. 2001), with mean fire intervals 50-100 years in the

Idaho Batholith (Arno and Petersen 1983) and over 300 years in Yellowstone (Morgan and Bunting 1999). In other subalpine forest types, historical fire rotations can be much longer. In mountain hemlock forests in the Cascade Range they can exceed 1,500 years (Franklin and Dyrness 1988, Lertzman and Krebs 1991, Agee 1993) and in the eastern Cascade Range and northern Rocky Mountains, *Larix lyallii* parkland rarely burns (Arno and Habeck 1972, Agee 1993).

Fires occur infrequently near treeline and are extremely rare in Krummholz and alpine tundra (Arno and Hammerly 1984, Benedict 2002, Körner 2003). In southern Colorado, only a fifth as much charcoal was found in lake sediments in the ATE as in subalpine forests (Anderson et al. 2008). Information for fire frequencies in alpine tundra is lacking. Although some research has been devoted to the effects of fire on alpine and Krummholz vegetation (Douglas and Ballard 1971, Potash and Agee 1998), studies of the frequency of fire in these systems is limited, likely because fires are either infrequent or absent (Malanson et al. 2007, Baker 2009). When fire does burn the ATE, tree reestablishment may not occur if facilitators have been removed (Billings 1969), seed sources are limited (Agee and Smith 1984, Little et al. 1994), or graminoids competitively exclude seedlings (Stahelin 1943). Thus, even though fires may be infrequent, their effects on the ecosystem can be long-lasting and profound. In contrast, fire frequency in non-forest areas in or adjacent to subalpine parkland may be similar to neighboring subalpine forests (Gabriel III 1976, Agee 1993, Baker 2009).

Longer fire seasons, drier fuels, and increased fire in adjacent continuous subalpine forests may increase area burned in the ATE. Conversely, barriers to fire spread, including meadows with higher fuel moisture and sparsely vegetated or barren areas, may buffer the ATE from disturbance in nearby forests. To determine if recent regional climate-driven increases in area burned are affecting ATEs, I conducted a geospatial analysis of wildfires over a period of 29 years (1984-2012) in subalpine parkland and alpine vegetation in the Pacific Northwest and Northern Rocky Mountains. I addressed four questions:

1. *How much area has burned in (1) subalpine parkland and (2) alpine vegetation during the study period, and how does that compare to published pre-settlement fire rotations?*
2. *Are the total areas burned in (1) subalpine parkland and (2) alpine vegetation correlated with total area burned across the broader landscape?*
3. *On a proportional basis, do (1) subalpine parkland and (2) alpine vegetation burn as much, more, or less than the broader landscape?*
4. *Was there a temporal trend in the proportion of area burned in (1) subalpine parkland and (2) alpine vegetation during the past three decades?*

I addressed each question for the entire area of study and in each of the eight ecoregions.

Methods

Study Area

The study area includes mountainous ecoregions within the Pacific Northwest and Northern Rocky Mountains of the U.S., including those in Oregon, Washington, Idaho, Montana, and Wyoming (Figure 1.1). I identified these as areas within the Level I Commission for Environmental Cooperation Ecoregion “Northwestern Forested Mountains,” and the Level II Ecoregion “Western Cordillera” (Commission for Environmental Cooperation 1997). Within the Level II Western Cordillera Ecoregion, the analysis was constrained to eight Level III Ecoregions, which had the majority of their area within the five northwestern and northern Rocky Mountain states (Figure 1.1).

From west to east, the study area comprises a gradient from maritime mesic to dry continental climates. Major high-elevation tree species west of the Cascade Range (Cascades and North Cascades) include *Abies lasiocarpa* (subalpine fir), *Tsuga mertensiana* (mountain hemlock), *Callitropsis nootkatensis* (Alaska cedar), and *Abies amabilis* (Pacific silver fir) (Arno and Hammerly 1984, Franklin and Dyrness 1988; nomenclature follows USDA NRCS 2015). East of the Cascade crest, high-elevation subalpine species include *A. lasiocarpa* and *Picea engelmannii* (Engelmann spruce), with *Larix lyallii* (subalpine larch) and *Pinus albicaulis* (whitebark pine) more prevalent near treeline (Arno and Hammerly 1984). East of the Continental Divide, in the Middle Rockies, *Pinus contorta* var. *latifolia* (lodgepole pine) is more common at high elevations (Arno and Hammerly 1984).

ATEs in all the ecoregions contain a variety of non-forest vegetation, including alpine tundra, alpine fellfields, shrub fields (particularly *Vaccinium* and heather species), and wet to dry meadows. Generally, wet meadows and shrub fields are more common in the Cascade Range; graminoids become more prevalent east of the Cascade Range and in the Rocky Mountains, and low-stature alpine tundra is more prevalent east of the Continental Divide. Despite this regional variation, there are many similarities among ecoregions in both forest and non-forest associations, particularly those from the eastern slopes of the Cascade Range to the western side of the Continental Divide (Ayres 1900, Daubenmire 1952, 1968, Gabriel III 1976, Franklin and Dyrness 1988).

ATEs vary in pattern and form among regions. In the western Cascades, ATEs tend to be more diffuse, showing a gradual reduction in tree height and density across the ecotone (Arno and Hammerly 1984, Harsch and Bader 2011). Heavy snow loads and a shorter snow-free season limit the maximum elevation of treeline and maintain treeless areas within subalpine parkland (Franklin and Dyrness 1988). East of the Cascade Crest, but west of the Continental Divide, ATEs vary in form, but “tree islands,” “ribbon forests” and open stands are prevalent. Here *P. albicaulis* establishes well in the open and sometimes facilitates other tree species (Callaway 1998, Resler et al. 2014). In the Middle Rockies and drier portions of the Canadian Rockies, ATEs tend to be more abrupt, with continuous forest directly adjacent to alpine vegetation although tree islands and ribbon forests are still common (Arno and Hammerly 1984). Within each ecoregion, however, a variety of treeline forms (*sensu* Harsch and Bader 2011)are present, reflecting differences in the abiotic limitations to tree growth at high elevation (*e.g.*, cold growing-season temperatures, short snow-free seasons, wind and associated snow redistribution, and lack of suitable substrates for growth) and differences in the strength of inter- and intra-specific interactions and the life stages that are affected. These differences in treeline form influence the distribution of fuels within the ATE, thus the probability of fire spread.

Geospatial data

I identified subalpine parkland and alpine vegetation using geospatial vegetation-type data, specifically the “Gap Analysis Landcover” layer from the U.S. Geological Survey (National Gap Analysis Program 2011). This layer models natural vegetation at a 30-meter resolution in hierarchical classes, and was derived from multi-sensor satellite imagery, digital elevation models, and topographical data sets (National Gap Analysis Program 2011). I created subalpine parkland and alpine vegetation layers based on the finest scale of vegetation described (Table 1.1). For this study, the alpine vegetation layer includes non-forest vegetation immediately adjacent to subalpine parkland, such as alpine shrub fields, as well as high alpine tundra distant from the nearest closed subalpine forest. Two ecoregions—Eastern Cascades and Columbia Mountains—had relatively little parkland and very little (<1% of the area) high-elevation non-forest vegetation (Table 1.2). Therefore, I did not include the alpine vegetation in the statistical analyses of those regions.

The vegetation layers used in this study are conservative representations of two vegetation landcover classes in the alpine treeline. I excluded some vegetation classes (*i.e.*, mountain hemlock forests and montane grasslands) that may have occurred in the lower bands of some

ATEs so as to exclude closed subalpine and montane forests. I also excluded barren areas (ice, water, and rock), common at high elevations—areas that do not burn. Including them would have underestimated the proportion of area burned and overestimated the fire rotation. I used high-resolution (1- to 2-m) imagery in Google Earth Pro (Google Inc. 2013) to determine if any large areas of alpine tundra, meadow, or subalpine parkland were not captured in the subalpine parkland and alpine vegetation layers, and if any source vegetation classes that I had used to make these two vegetation layers included large areas of montane forest.

I obtained geospatial fire data from the "Monitoring Trends in Burn Severity" (MTBS) Program (Eidenshink et al. 2007, Monitoring Trends in Burn Severity 2014). Data were used to calculate area burned across all vegetation types and within subalpine parkland and alpine vegetation for each year of the study period (1984-2012). MTBS data include all fires >400 ha and are derived from fire perimeters from federal and state fire databases and a Landsat-derived remotely sensed measure of burn severity, the differenced Normalized Burn Severity Ratio (dNBR). DNBR is computed as the change from pre- to post-fire in the surface spectral reflectance of the near and mid-infrared bands of Landsat satellite imagery (Key 2006). It has been correlated with burn severity and tree mortality in the field in the Pacific Northwest and Rocky Mountains (Cansler and McKenzie 2012, Parks et al. 2014).

I do not use the MTBS data set to assess severity, but to calculate the area burned. Because some fire perimeters may include areas that did not burn, I calculate the annual area burned in two ways: (1) the area classified in the MTBS burn-severity data as anything other than "unburned to low" (*i.e.,* sum of low, moderate, high, increased greenness, and unclassified; Eidenshink et al. 2007), and (2) the full area within a fire perimeter (*i.e.,* all severity classes). I report results based on the former, but include results based on the latter for comparison (in the appendix). Excluding the "unburned to low" should estimate the true area burned more accurately because it excludes large unburned areas that are often included in fire perimeters (Kolden et al. 2012, 2015b). Errors of inclusion due to inaccurate fire perimeters may be higher for ATEs because fire perimeters are often extended to the nearest major topographical break, such as ridgetops. Thus, it is likely that perimeters include unburned wet and barren alpine areas. Conversely, excluding the "unburned to low" class may exclude areas that burned at severities too low to detect by satellite (Key 2006, Kolden et al. 2012, 2015b, Cansler and McKenzie 2012). Moreover, assessments of classification accuracy of dNBR data show that burned areas occur in the "unburned to low class" and unburned areas occur in all the other classes (Cansler and McKenzie 2012, Parks et al. 2014). Misclassification may be exacerbated by the MTBS classification procedures, which are determined by a remote sensing specialist after visual evaluation of the burn severity images, as opposed to empirical relationships with field data (Kolden et al. 2015b). By also reporting estimates of area burned based on fire perimeters, I decrease the uncertainty due to misclassification and inaccurate perimeters, but some inaccuracies remain.

Statistical Analyses

For each analysis, I summarized results for eight ecoregions combined (hereafter "study area") and individually to assess variation among them. Analyses were limited to ecoregions in which subalpine or alpine vegetation made up ≥ 0.1% of the landscape.

For subalpine parkland and alpine vegetation within the study area and each ecoregion, I calculated the total area, total area burned, proportion of area burned on an annual and 29-year basis. I also wanted to be able to compare burning during the study period with published fire rotations prior to Euro-American settlement. Therefore I calculated the fire rotation—the time needed to burn an area equal to each analysis area (Agee 1993)—based on the 29-year study period. The fire rotations derived from the 29-year study period should be viewed as snapshots of current conditions, but due to the short analysis period, should not interpreted as indicative of the long-term fire regimes of these ecosystems (Agee 1993).

I calculated the Spearman's rank correlation between annual area burned in each vegetation type (subalpine parkland and alpine parkland) and the annual area burned across the broader landscape (all vegetation classes; hereafter "total area burned"; n = 29). I used Spearman's rank correlation because the ranges of fire size were not normally distributed. If similar climate factors influence area burned in subalpine parkland and alpine vegetation as in the broader landscape, they should be strongly correlated. Correlations should also be stronger in ecoregions where the proportion of the landscape in subalpine parkland or alpine vegetation are larger, such as for subalpine parkland in the North Cascades (9.5%) and alpine vegetation in the Middle Rockies (5.1%) than in ecoregions with low cover of these vegetation types (Table 1.2). A strong correlation would also suggest that future increases in area burned should lead to similar increases in the subalpine and alpine.

In a complementary analysis, I used simple linear regression to assess if the total area burned annually was a significant predictor of area burned in subalpine parkland or alpine vegetation, allowing me to compare the slopes and amount of variance explained by the regressions between ecoregions. I log-transformed ($\log(1 + x)$) all area data before regression analyses, to stabilize the variance, and assessed whether data met the assumptions of regression using standard methods (*e.g.*, normal probability plots, residual plots, and partial residual plots; Kutner et al. 2005).

For these analyses, the total area burned includes burned subalpine parkland and alpine vegetation. Although this could lead to non-independence, the correlation between the two should be small because, for most ecoregions, subalpine parkland and alpine vegetation cover <2% of the ecoregion (Table 1.2). Alpine vegetation covers 5.1% of the Middle Rockies and subalpine parkland covers 9.5% of the North Cascades.

I compared two alternative hypotheses: (1) subalpine and alpine burn less frequently than the broader landscape, or (2) the subalpine and alpine burn when the broader landscape burned. The first implies that despite increases in the overall area burned since the mid-1980s (Littell et al. 2009), limited fuel connectivity, shorter fire seasons, and differences in microclimate with elevation in still limit burning in the ATEs. The latter implies that differences in fuels and local climate between the ATE and the broader landscape are relatively unimportant. I tested if the proportion of area burned annually in subalpine parkland and alpine vegetation was different from the proportion of area burned annually across all vegetation classes. First, I directly compared the proportions of subalpine parkland and alpine vegetation burned with the proportion of the total area burned for each analysis area that I calculated for the full study period. Differences between expected (*i.e.*, the proportion of the broader landscape that burned; Cumming 2001, Podur and Martell 2009) and observed proportion of area burned in subalpine parkland or alpine vegetation should be viewed as real differences. Second, using individual years as samples (n = 29), I tested if the proportion of area burned in subalpine parkland and alpine vegetation was different from the proportion of annual area burned across all vegetation classes. I used the Wilcoxon Signed Rank test with the two-tailed hypothesis that the observed area burned did not differ from that expected, with each year as a sample (α = 0.05, n = 29).

To determine if there was a temporal trend in the proportion of area burned in subalpine parkland or alpine vegetation during the past three decades, I tested the linear relationship between log-transformed area burned and year (α = 0.05). Separate regressions were tested for the study area and for each ecoregion. Because the time series is short (29 years), results are interpreted with caution: the sample size may be too small to detect a weak trend and the deviation due to a single year may influence results. All statistical tests were conducted in the statistical program R (R Core Team 2014).

Results

Subalpine parkland made up 1.2% of the full study area (784,193 ha; Table 1.2) and 7% of it burned during the study period (Table 1.3). Alpine vegetation made up 1.6% of the area (1,027,680 ha) and 3% during the study period. As expected, ecoregions with greater representation of subalpine parkland and alpine vegetation (Table 1.2) usually had higher proportions burned (Table 1.3). An exception was the Middle Rockies, where a relatively small percentage (3%) of the alpine vegetation burned despite making up >5% of the area. Regions with large proportions of area burned also had large proportions subalpine parkland burn. In the Blue Mountains, 19% of the alpine vegetation burned, reflecting a high proportion of total area burned (11%), even though the alpine covered only 0.1% of the landscape (Table 1.2). Likewise, the large proportion of subalpine parkland burned in the Idaho Batholith (22%) probably due to the notably higher total area burned (29%).

There were strong positive correlations between area burned in subalpine parkland or alpine and total area burned for the full study area ($\rho = 0.90$, $P < 0.0001$ and $\rho = 0.887$, $P < 0.001$, respectively; Table 1.4). Among individual ecoregions, relationships were moderately to strongly positive ($\rho = 0.61$ to $\rho = 0.98$, all $P < 0.001$; Table 1.4). Correlations were particularly strong in large-fire years (Figure 1.2).

Linear regressions predicting area burned in the subalpine or alpine from total area burned were significant for all but one ecoregion ($P < 0.01$; Table 1.4). For the full study area, models explained 84% and 76% of the variance in area burned in the subalpine and alpine, respectively (P < 0.001). For individual ecoregions, significant models explained 28% to 88% of the variance in area burned. Greater variation was explained, and slopes were generally steeper, in ecoregions where more area burned, such as the Canadian Rockies and Idaho Batholith.

For the full study area and study period, the proportion of subalpine parkland burned was slightly less than the proportion of the entire landscape burned (7% and 8% respectively). For the full study area on an annual basis the proportion of subalpine parkland burned differed significantly (P = 0.031; Table 1.5). However, proportions did not differ when estimates of area burned were based on the total area within fire perimeters (Appendix 1.1). The proportion of subalpine parkland burned exceeded the proportion of total area burned in some years, particularly when a large area burned (Figure 1.3).

I observed considerable variation in patterns of burning among ecoregions. Over the 29-year study period, a larger proportion of subalpine parkland burned than the broader landscape in the Canadian Rockies, Cascades, Columbia Mountains, and Middle Rockies (Table 1.3). Likewise, annually, the proportions of subalpine parkland and total area burned did not differ in the Canadian Rockies, Cascades, and Columbia Mountains (Table 1.5). Over the whole study period and annually, the proportions of subalpine parkland burned were lower in the other five ecoregions (Table 1.3, Table 1.5).

Across the entire study area, and in all ecoregions except the Blue Mountains, the proportion of alpine vegetation that burned was smaller than the proportion of the broader landscape that burned, over the 29-year study period (Table 1.3) and on an annual basis (Table 1.5). In the Blue Mountains, more (19%) of the alpine vegetation burned during the 29-year study period, than the broader landscape (11%), but differences were not significant when tested with annual data (P =

0.142).

I did not detect a temporal trend in the proportion of area burned over the study period, with the exception of an increase in the total area burned in the Idaho Batholith ($P < 0.001$; other results not shown).

Statistical analyses based on area within fire perimeters (rather than the summation of burn severity classes) rarely changed statistical outcomes. However, it did result in no significant difference ($P = 0.100$) between the proportion of subalpine parkland burned and the proportion of the entire study area burned (Table 1.5; App. 1 Table 1.3); differences were significant ($P = 0.031$) when area burned was calculated based on fire severity definitions. In some cases, even when estimates of area burned differed greatly, model outcomes did not change. In the most extreme case, including areas rated as “unburned to low severity” more than doubled the alpine area burned in the Middle Rockies (23,469 vs. 58,644 ha). Without field validation, particularly in alpine, we cannot determine which estimate are more accurate.

Discussion

This study provides one of the first regional-scale assessments of area burned that specifically focuses on the alpine treeline ecotone (ATE). Other studies using geospatial approaches have assessed area burned at regional scales in the western US and its connection to climate (Littell et al. 2009, Abatzoglou and Kolden 2013), fire management, and forest type (Miller et al. 2012, Mallek et al. 2013). Most previous research on fire regimes in high-elevation forests and the alpine treeline ecotone is based on dendrochronological methods. These provide a long temporal record of the mean and variation in fire frequency, but usually do not summarize recent trends, and may be difficult to scale up without many field sites. This study bridges the gap between large-scale analysis of area burned across many vegetation types and previous site-based work on fire regimes in the subalpine and alpine treeline ecotones, by providing information on a specific ecological community, while conducting analyses at two relatively large extents, that of ecoregions and the Pacific Northwest and northern Rocky Mountains.

The most surprising result of this study was that on a proportional basis the area of subalpine parkland burned was either comparable or greater than the broader landscape in four of the eight ecoregions: the Cascades, and three of the four Rocky Mountain ecoregions (Canadian Rockies, Columbia Mountains, and Middle Rockies). Prior to Euro-American settlement subalpine parkland generally had longer fire rotations than other forest types (175-350 years in the Northern Rockies and Interior Northwest; see summaries in Agee 1993, Baker 2009). Thus I expected that it would have burned less than forest types with shorter fire rotations, which are usually considered to be more flammable.

There are several possible explanations for this counterintuitive result: (1) changes in the climate, (2) previous fire exclusion in lower-elevation forest types, (3) managers allowing more fires to burn for resource benefit (*i.e.,* increased “wildland fire use”), (4) increased ignitions, and (5) the small area burned across the broader landscape compared to the pre-settlement period.

Observed changes in climate in the study area, such as increased mean annual temperature, decreased summer and autumn precipitation, reduced snowpack, and earlier snowmelt (Mote et al. 2005, Abatzoglou et al. 2014, Johnstone and Mantua 2014, Jolly et al. 2015) all increase the length of the fire season. Higher rates of warming at high elevations, as has been observed worldwide (Pepin et al. 2015), could provide one explanation for the results. In the western U.S., though, positive trends in the minimum annual temperature are slightly greater at high elevations than low elevations, but these differences are not statistically significant (Oyler et al. 2015). On the other hand, even though the climate has changed similarly at high and low elevations, the effects of those changes on length of the fire season and fuel flammability could differ between elevations. The flammability of subalpine parkland may be increasing more rapidly than lower elevations due to longer snow-free seasons, and drier greater continuity of surface fuels. Climate and its interactions with other disturbances (*e.g.*, bark beetles) could increase standing and down dead fuel loadings in the ATE through increased tree mortality. Climate may also contribute to fuel accumulation in the ATE in the form of greater tree density in meadows (Franklin et al. 1971, Rochefort and Peterson 1996), which could, theoretically, enhance fire spread in the ATE by increasing the amount and connectivity of fuels (Schwartz et al. 2015). Nevertheless, increased density of trees in the ATE is unlikely to be important, as the analysis of field data in Chapter 2 shows that small trees are less likely to burn than large tree in the ATE, probably due to higher live fuel moisture, higher relative humidity close to the ground, decreased wind speeds, or some combination thereof.

A second indirect climatic mechanism is that increased flammability in subalpine forests or neighboring forests types could be driving area burned in subalpine parkland. Continuous subalpine forests adjacent to subalpine parkland have had relatively larger increases in area burned than other forest types. From 1970-2003, the largest increase in large-fire frequency occurred in mid- and high elevations (1680-2590 m) across the western United States, and in the northern Rocky Mountains region (Westerling et al. 2006). Similar rapid changes in fire regime have also been observed at smaller spatial scales. In the northern Rocky Mountains, more subalpine forest burned than did mid-montane forest from 1984-2010 (19% vs. 12% based on data in Harvey 2015). Moreover, in two of three subalpine forest types, the mean annual area burned between 1984 and 2010 exceeded that of historical fire regimes (Zhao et al. 2015). In eastern Washington, Oregon, and Northern California, subalpine forests had the greatest mortality in forest inventory plots of any forested type: approximately 30% of plots in subalpine forests had very high mortality rates (≥25%/year), likely due to fire (Reilly 2014). In the Northern Cascade Range of Washington, relationships between climate and area burned and between fire severity and patch size were more pronounced in cooler and drier subalpine forests than in warmer and drier forests or cooler and wetter forests (Cansler and McKenzie 2014). Historically, fires in montane and subalpine forests have been periodic, but widespread when climate is conducive to burning (Kipfmueller 2003). Fire regimes in these forests may be more responsive to climate change because fuels are more continuous and could support extreme fire behaviour and rapid growth (Bessie and Johnson 1995, Cansler and McKenzie 2014). Because fire is a contagious process, increased exposure should cause fire to increase in a non-linear fashion in rare vegetation types (Kennedy and McKenzie 2010), such as subalpine parkland and alpine vegetation.

In addition to climate, past fire suppression may also be contributing to high amounts of area burned in neighboring forests. Fire suppression did not likely contribute to an increase of fuels above the historical range of variability in subalpine parkland, because these areas had longer fire rotations than the period of human suppression of fire. However, suppression efforts have contributed to increased contagion across the landscape (Hessburg et al. 2005, 2015, Miller and Safford 2012, Collins et al. 2015), which may contribute to increase fire spread into subalpine parkland

Changes in fire management may also have contributed to area burn in subalpine parkland in the three Rocky Mountain ecoregions. Each of these ecoregions have wildland fire use programs through which natural fires are not suppressed, but allowed to burn. Areas in which wildland fire use is allowed—national wilderness areas and national parks—include a disproportionate amount of subalpine and alpine vegetation relative to other land designations (Scott et al. 2001, Dietz et al. 2015), making it more likely that those vegetation types will burn. Moreover, even when high elevation fires are not officially allowed to burn, they may be less aggressively suppressed than fires in low-elevation forests that are near human habitation.

It is also possible that lightning ignitions have increased in subalpine parkland relative to the broader landscape in the Cascade and Rocky Mountain ecoregions. Increased ignitions could be due to changes in the spatial locations of lightning, and decreases in fuel moisture related to changes in climate. Given the focus on area burned and use of data from only larger fires (and not the number of fires) landscape flammability should have a much stronger influence on my results than frequency of lightning strikes that successfully ignite fires. Nevertheless, I cannot rule out the possibility that changes in the frequency of lightning strikes and ignition patterns contribute to my results. An analysis of temporal trends in ignitions and the number of fires within different forest types could elucidate the relative importance of changes in the location of ignition sources, climate-driven changes in ignition probabilities within a subalpine parkland (or other focal vegetation types), and fire spread from other vegetation types.

Finally, subalpine parklands may be burning as often the broader landscape because historically frequent-fire forests are now burning less frequently. In other words, the statistically indistinguishable differences in area burned might be driven by fire suppression in low-elevation forests, not more area burned in subalpine forests. Patterns for the Idaho Batholith lend to this idea, by showing there are no significant differences when a greater proportion of the broader landscape burns. Within this ecoregion, fires are allowed to burn for resource benefit within the Selway-Bitterroot and Frank Church-River of No Return Wildernesses. Subalpine parkland in the Idaho Batholith burn significantly less than the landscape, even though much more of the subalpine parkland burned (22%) than in any other ecoregion. The differences were not significant because much more of the broader landscape burned as well (29%).

Most fire rotations remain within the bounds of historical estimates. Fire rotations (Table 1.3) for subalpine parklands in the Blue Mountains, Canadian Rockies, and Middle Rockies were comparable to the 175- to 350-year range previously described for the Rocky Mountains (see summaries in Baker 2009). However, fire rotations were shorter in the Idaho Batholith (91-132 years). Reconstructions of mean fire frequencies at two sites in this ecoregion were between 50-80 years (Arno and Petersen 1983), indicating fires historically burned more frequently than in other ecoregions in the Northern Rocky Mountains.

Fire rotations in the Cascades, North Cascades, and Eastern Cascades were longer than published ranges for *A. lasiocarpa* and *P. albicaulis.* Current and historical fire rotations in the Cascades and North Cascades ecoregions are more difficult to compare, because my results and the original geospatial data layers aggregate multiple forest types with very different fire rotations. For example, *L. lyallii and T. mertensiana* may have fire rotations >1000 years (Lertzman and Krebs 1991, Agee 1993), although some *L. lyallii* stands burn more often (Arno and Habeck 1972). In contrast, rotations in *A. lasiocarpa* and *P. albicaulis* forests are usually much shorter (<250 years; Fahnestock 1976, Franklin and Dyrness 1988, Agee et al. 1990). I excluded *T. mertensiana* forests from this analysis (see Methods), but *L. lyallii* stands were captured within the subalpine parkland, and are a major subalpine and treeline forest type in the North Cascades, where they form large pure stands and occupy a variety of habitats (Arno and Habeck 1972). Whether the longer fire rotations reflect the inclusion of less flammable *L. lyallii* stands (Arno and Habeck 1972, Agee 1993) is unclear. The longer fire rotations in the Cascade Range than in the northern Rocky Mountains could also be due to other factors that affect responsiveness to changes in climate (Abatzoglou and Kolden 2013, Kolden et al. 2015a). These include a cooler wetter climate, more unburnable area, and topographical barriers to fire spread (Cansler and McKenzie 2014), as well as a less extensive wildland fire-use program (Cansler 2011).

In the alpine, fire rotations were very long, exceeding 1,000 years in six of the eight ecoregions. Although past studies indicate that alpine vegetation rarely burns, and my results correspond with these observations, the small sample period makes it unlikely that I captured rare burning events. The proportion of alpine or subalpine vegetation in each ecoregion, and the area that burned across all vegetation types during the short (29-year) time period of the study, affected the probability of alpine and subalpine vegetation burning during the study period. Because alpine vegetation usually occupies a small portion of the landscape (<1%, except in the Central Rockies), fire rotations could shift with a few consecutive active or subdued fire years. For example, the Blue Mountains, Columbia Mountains, and Eastern Cascades—which had proportionally low cover of subalpine and alpine vegetation—proportional area burned was either very high or very low, and the fire rotations were very short or very long, respectively.

Conclusions

Increased fire frequencies may have amplifying or stabilizing feedbacks (*sensu* Chapin et al. 2009) on climate-driven changes in the ATEs. Increased fire in these ecosystems could hasten climate-driven changes, by removing cold-adapted and alpine species at the margins of their ranges (Lesica and McCune 2004, Gottfried et al. 2012), and by creating growing space that allows lower-elevation species to establish and spread. Conversely, increased fire could reduce tree cover, thereby maintaining non-forest vegetation, and counteract ongoing responses to climate change, such as upward movement of treelines (Brubaker 1986, Harsch et al. 2009), and tree invasion into subalpine meadows (Franklin et al. 1971, Taylor 1995, Rochefort and Peterson 1996). In these contexts, fire could provide secondary benefits to wildlife species that use or depend on high-elevation non-forest vegetation. Fire may also interact with other stresses and disturbances, which may maintain open areas for long periods. For example, by changing patterns of snow deposition (Billings 1969), fire increased tree mortality and permanently converted ribbon forest to a snow-maintained unforested state. Likewise, by removing anchor points, such as standing trees, that stabilize snowpack, fire can increase the frequency and magnitude of avalanches, thus maintaining the diverse flora and fauna found in these habitats (Krajick 1998, Germain et al. 2005, Bebi et al. 2009).

Research has projected that climate change will increase area burned in western North America (Flannigan et al. 2006, Littell et al. 2010, Jolly et al. 2015). To plan for and adapt to the impacts of climate change in subalpine parklands, more research is needed on the direct effects of fire on vegetation structure and species diversity, the indirect impacts of fire on wildlife, soils, snow hydrology, and other processes, and subsequent feedbacks on vegetation. Until sufficient area burns and fire becomes self-limiting, fire will remain an important process in subalpine parkland, and an infrequent, but consequential, process in alpine vegetation.

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Tables

Table 1.1. Vegetation classes from the Gap Analysis landcover data used to identify subalpine parkland and alpine vegetation for this analysis. Data had a 30-meter resolution. Area totals are for the entire study area (the eight Level III ecoregions analyzed).

Vegetation class	Area (ha)	Level 1 class	Level 2 class	Level 3 class
Subalpine parkland	493,138	Forest and woodland	Conifer dominated forest and woodland (xeric-mesic)	Northern Rocky Mountain subalpine woodland and parkland
Subalpine parkland	92,930	Forest and woodland	Conifer dominated forest and woodland (xeric-mesic)	Rocky Mountain subalpine-montane limber-bristlecone pine woodland
Subalpine parkland	209,141	Forest and woodland	Conifer dominated forest and woodland (mesic-wet)	North Pacific maritime mesic subalpine parkland
Alpine	21,645	Shrubland, steppe and savanna	Alpine and avalanche chute shrubland	North Pacific dry and mesic alpine dwarf-shrubland, fell-field and meadow
Alpine	114,146	Shrubland, steppe and savanna	Alpine and avalanche chute shrubland	Rocky Mountain alpine dwarf-shrubland
Alpine	24,071	Shrubland, steppe and savanna	Alpine and avalanche chute shrubland	Rocky Mountain alpine tundra/fell-field/dwarf-shrub
Alpine	210,269	Grassland	Alpine grassland	Rocky Mountain alpine fell-field
Alpine	609,469	Grassland	Alpine grassland	Rocky Mountain dry tundra
Alpine	49,309	Grassland	Alpine grassland	North Pacific alpine and subalpine dry grassland

Table 1.2. Area of subalpine parkland and alpine vegetation within each ecoregion. Ecoregions are shown in Figure 1.

Ecoregion	Total (ha)	Subalpine (ha)	Alpine (ha)	Proportion in subalpine parkland	Proportion in alpine vegetation
Study Area	62,946,106	784,193	1,027,680	0.012	0.016
Blue Mountains	7,091,151	42,918	5,099	0.006	0.001
Canadian Rockies	5,693,431	109,612	41,864	0.019	0.007
Cascades	4,643,400	84,913	21,346	0.018	0.005
Columbia Mountains/ Northern Rockies [a]	13,744,447	22,116	102	0.002	0.000
Eastern Cascades Slopes and Foothills [b]	5,617,714	8,294	1,834	0.001	0.000
Idaho Batholith	6,028,341	77,293	43,274	0.013	0.007
Middle Rockies	16,446,161	89,754	842,103	0.005	0.051
North Cascades	3,681,462	349,293	72,058	0.095	0.020

[a] Hereafter referred to as Columbia Mountains, to prevent confusion with the larger northern Rocky Mountain region. I use the latter to describe the combined Canadian Rockies, Columbia Mountains, Idaho Batholith, and Middle Rockies ecoregions.
[b] Hereafter referred to as Eastern Cascades

Table 1.3. Total area burned over the 29-year study period within vegetation groups in each ecoregion, and the proportion of each vegetation type that burned. The fire rotation was calculated as 1/(mean annual proportion of area burned), and represents the number of years it would take to burn an area equal to the area of landcover for a given vegetation type.

Ecoregion	Area burned (ha)			Proportion burned			Fire rotation (years)		
	Subalpine parkland	Alpine vegetation	Total area	Subalpine parkland	Alpine vegetation	Total area	Subalpine parkland	Alpine vegetation	Total area
Study Area	55,137	27,510	5,015,686	0.070	0.027	0.080	412	1,083	364
Blue Mountains	3,722	942	769,493	0.087	0.185	0.109	334	157	267
Canadian Rockies	8,863	531	317,990	0.081	0.013	0.056	359	2,288	519
Cascades	3,268	381	140,947	0.038	0.018	0.030	754	1,626	955
Columbia Mountains	695	0	240,568	0.031	0.004	0.018	923	6,583	1,657
Eastern Cascades	211	24	303,403	0.025	0.013	0.054	1,138	2,180	537
Idaho Batholith	17,013	1,525	1,757,879	0.220	0.035	0.292	132	823	99
Middle Rockies	7,162	23,469	1,191,033	0.080	0.028	0.072	363	1,041	400
North Cascades	14,201	638	294,374	0.041	0.009	0.080	713	3,278	363

Table 1.4. Spearman's rank correlations (ρ) between annual area burned within each ecoregion and the area of subalpine parkland or alpine vegetation burned (n=29) [a], and results of linear regressions predicting annual area of subalpine parkland or alpine vegetation burned as a function of annual total area (all vegetation types) burned (n = 29). Data were log-transformed prior to analysis.

Ecoregion	Subalpine parkland						Alpine Vegetation					
	ρ [a]	Intercept	Slope	t	P	R^2	ρ [a]	Intercept	Slope	t	P	R^2
Study Area	0.90	-10.17	1.44	7.31	<0.001	0.84	0.87	-10.57	1.34	6.90	<0.001	0.73
Blue Mountains	0.72	-3.76	0.65	4.76	<0.001	0.32	0.67	-2.41	0.44	3.24	0.001	0.28
Canadian Rockies	0.98	-0.23	0.58	8.33	<0.001	0.85	0.87	-0.23	0.28	4.12	<0.001	0.55
Cascades	0.79	0.00	0.42	5.85	<0.001	0.55	0.87	-0.11	0.30	4.22	<0.001	0.67
Columbia Mountains [b]	0.61	-0.65	0.26	2.93	0.004	0.28	…	…	…	…	…	…
Eastern Cascades [b]	0.65	-0.54	0.17	1.67	0.096	0.14	…	…	…	…	…	…
Idaho Batholith	0.93	-3.51	0.83	6.74	<0.001	0.68	0.76	-2.85	0.48	3.94	0.002	0.35
Middle Rockies	0.88	-2.77	0.63	5.66	<0.001	0.47	0.85	-3.15	0.72	6.58	<0.001	0.45
North Cascades	0.80	-0.46	0.61	7.78	<0.001	0.62	0.77	-0.53	0.24	3.06	<0.001	0.33

[a] All correlations were significant (*P < 0.001*).

[b] Alpine vegetation classes in the Columbia Mountains and Eastern Cascades Ecoregions were not evaluated because there was little alpine vegetation in those ecoregions.

Table 1.5. Wilcoxon Signed Rank Test comparing the annual proportion of subalpine parkland or alpine vegetation burned and the annual proportion all vegetation types burned. Significant results with negative median support the hypothesis that subalpine parkland or alpine vegetation is less likely to burn than the broader landscape (there were no tests with a significant result and a positive median suggesting greater likelihood of burning than the broader landscape).

Ecoregion	Subalpine Parkland			Alpine Vegetation		
	V	*P*	Estimated Median	*V*	*P*	Estimated Median
Study area	117	0.031	-0.0003	26	0.000	-0.0013
Blue Mountains	86	0.008	-0.0010	138	0.142	-0.0006
Canadian Rockies	102	0.237	0.0004	2	0.000	-0.0010
Cascades	58	0.623	-0.0003	18	0.010	-0.0007
Columbia Mountains [a]	76	0.036	-0.0002	...	...	...
Eastern Cascades [a]	47	0.001	-0.0010	...	...	...
Idaho Batholith	84	0.007	-0.0011	0	0.000	-0.0041
Middle Rockies	73	0.006	-0.0003	0	0.000	-0.0009
North Cascades	50	0.024	-0.0011	0	0.000	-0.0024

[a] Alpine vegetation classes in the Columbia Mountains and Eastern Cascades Ecoregions were not evaluated due to low cover.

Figures

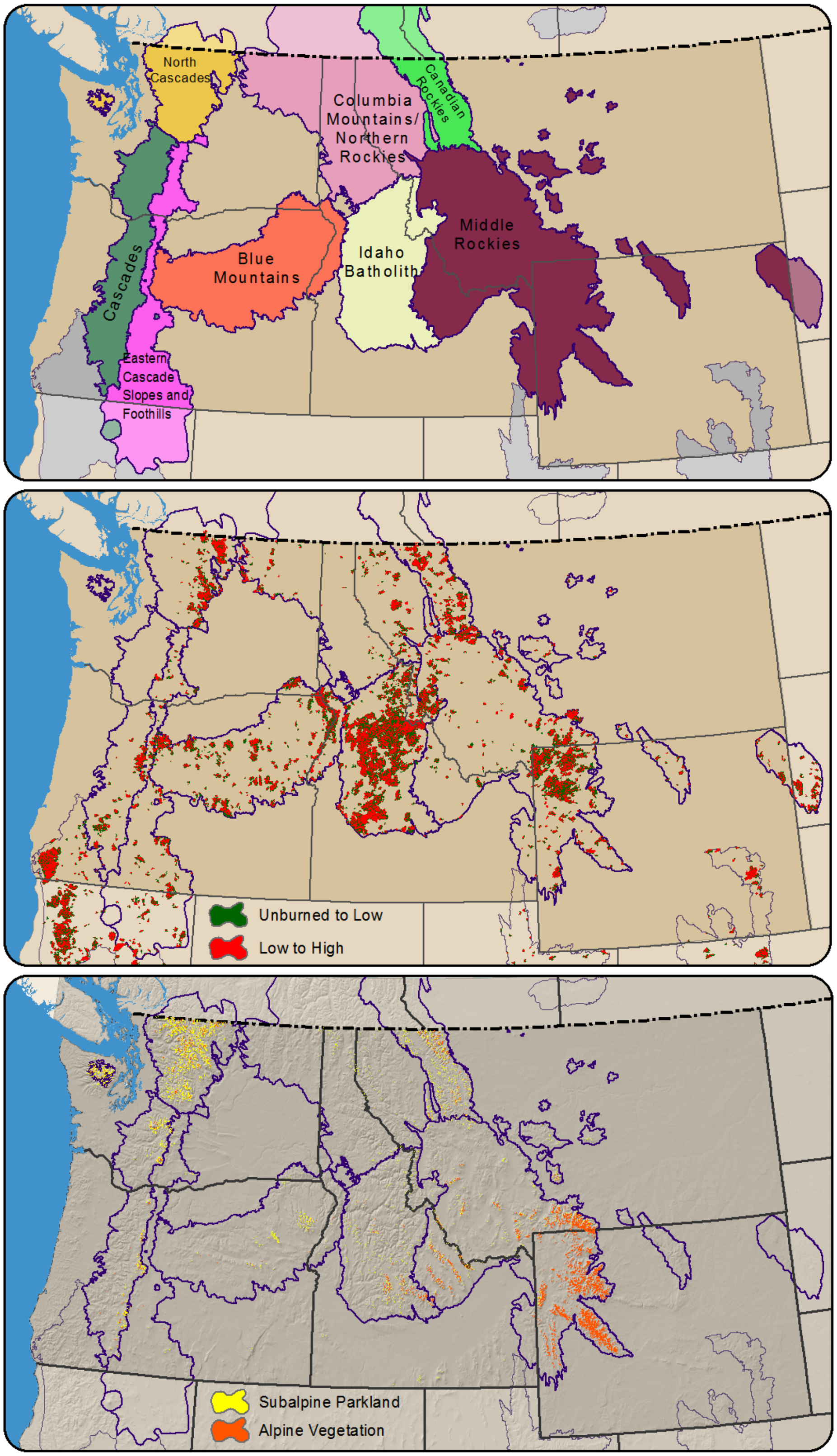
North Cascades
Columbia Mountains/ Northern Rockies
Canadian Rockies
Cascades
Blue Mountains
Idaho Batholith
Middle Rockies
Eastern Cascade Slopes and Foothills
Unburned to Low
Low to High
Subalpine Parkland
Alpine Vegetation

Figure 1.1. Map of study area.

Top: Level III CEC Ecoregions included in the analysis. These Level III Ecoregions were within “Western Cordillera” Level II Terrestrial Ecoregion (shaded grey), and had the majority of their area within the five Pacific Northwest and northern Rocky Mountains states: Oregon, Washington, Idaho, Montana, and Wyoming. Middle: classified burn severity images for all fires >400 ha from 1984-2012 within the “Western Cordillera” CEC Level II Terrestrial Ecoregion. Burn severity is shown by color: unburned to low = dark green; all other classes (low, moderate, high, enhanced regrowth, and unclassified) = red. Bottom: Subalpine (yellow) and alpine (orange) vegetation classes.

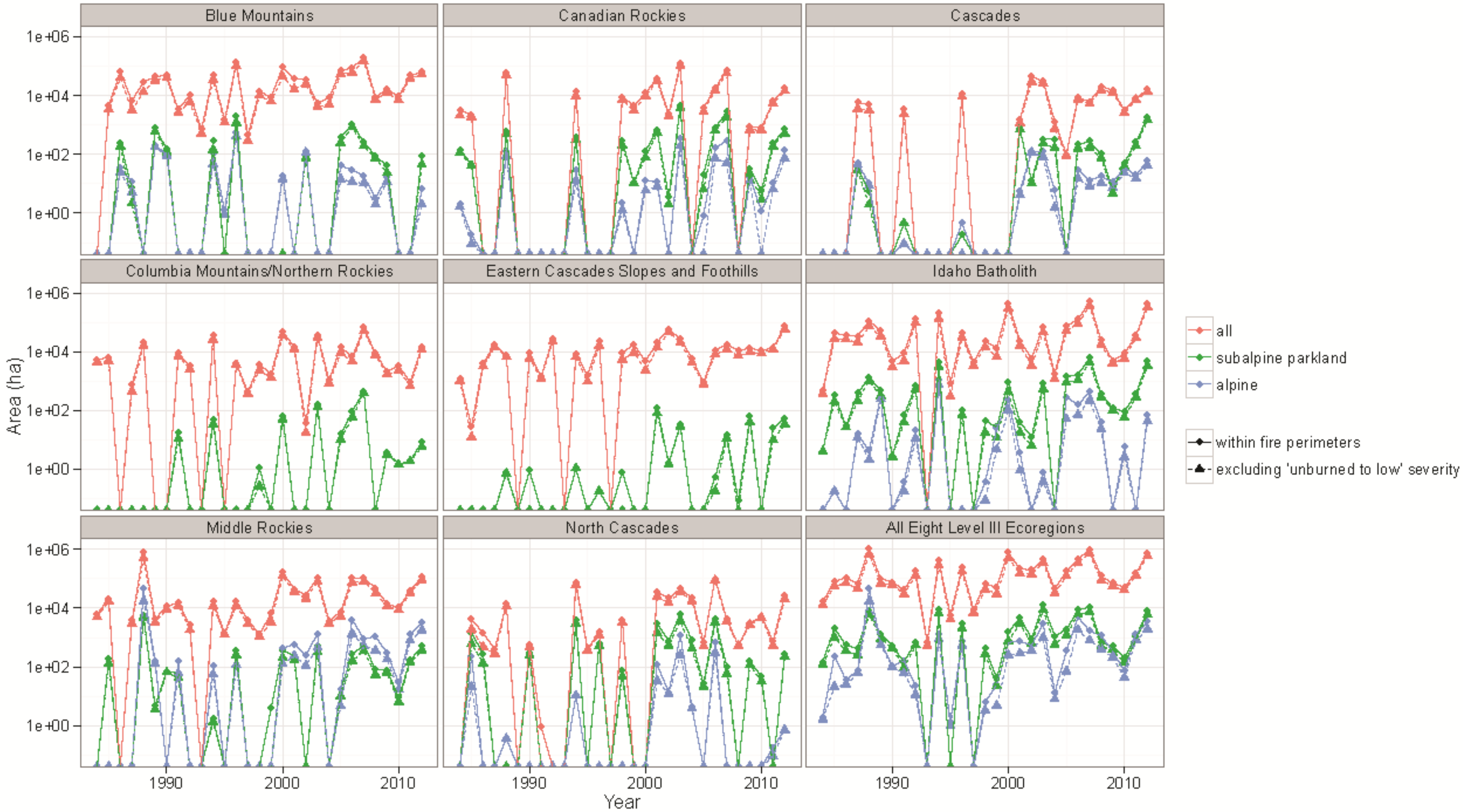

Figure 1.2. Time series of area burned (total, subalpine parkland, and alpine vegetation) for each ecoregion, and across the entire study area (bottom right panel).

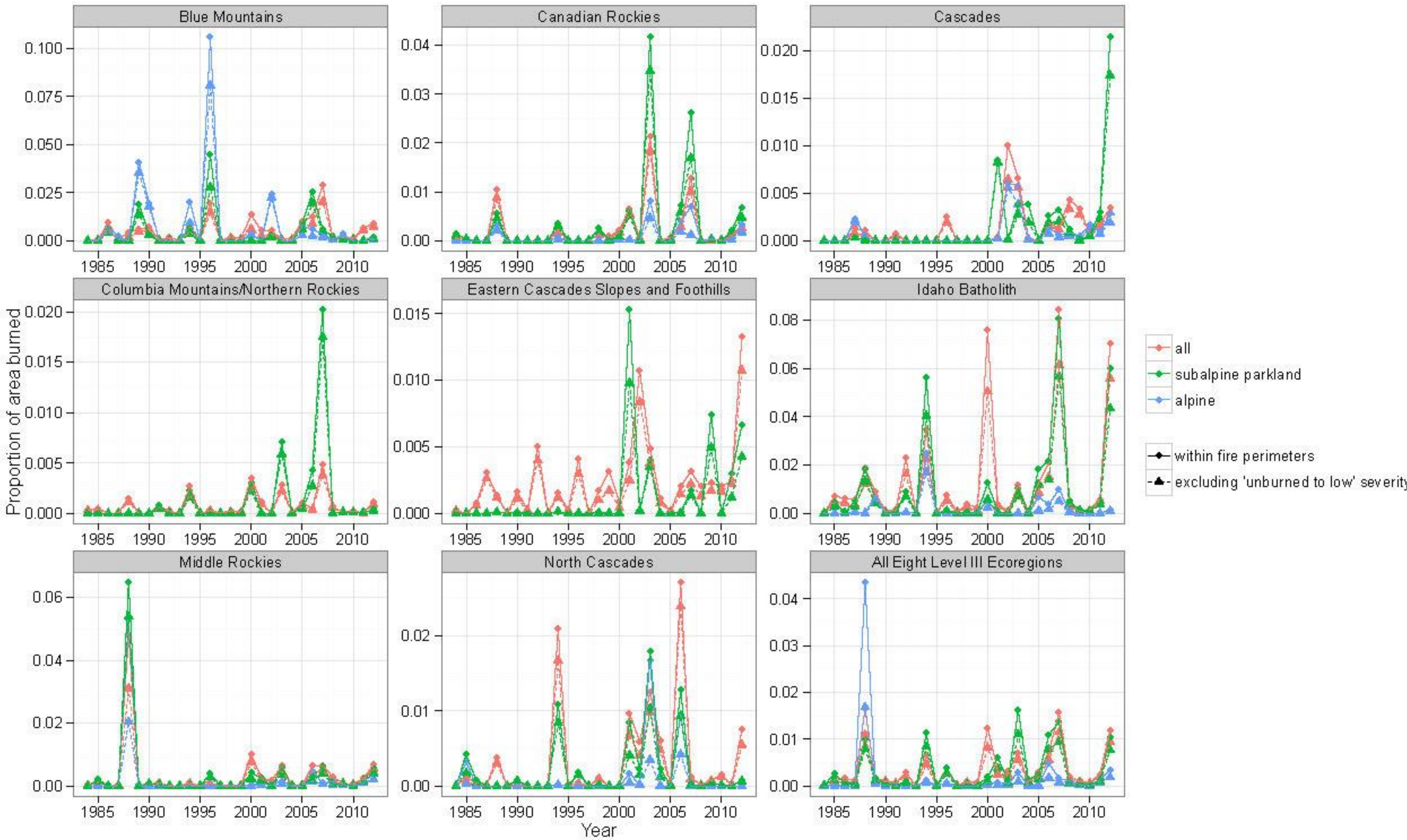

Figure 1.3. Time series of the proportion of area burned (total, subalpine parkland, and alpine vegetation) for each ecoregion, and across the entire study area (bottom right panel).

Appendix 1.1. Results when the analyses is performed using the areas within fire perimeters

Table A1.1. Total area burned over the 29-year study period within vegetation groups in each ecoregion, and the proportion of each vegetation type that burned. The fire rotation was calculated as 1/(mean annual proportion of area burned), and represents the number of years it would take to burn an area equal to the area of landcover for a given vegetation type.

Ecoregion	Area burned (ha)			Proportion burned			Fire rotation (years)		
	Subalpine parkland	Alpine vegetation	Total area	Subalpine parkland	Alpine vegetation	Total area	Subalpine parkland	Alpine vegetation	Total area
Study Area	78,621	66,565	6,965,218	0.100	0.065	0.111	289	448	262
Blue Mountains	5,467	1,255	1,120,291	0.127	0.246	0.158	228	416	184
Canadian Rockies	11,524	1,156	385,028	0.105	0.028	0.068	276	468	429
Cascades	4,160	512	184,418	0.049	0.024	0.040	592	4,114	730
Columbia Mountains	865	1	313,346	0.039	0.007	0.023	742	1,050	1,272
Eastern Cascades	322	41	403,307	0.039	0.023	0.072	748	919	404
Idaho Batholith	24,686	2,681	2,496,174	0.319	0.062	0.414	91	1,208	70
Middle Rockies	9,307	58,644	1,691,822	0.104	0.070	0.103	280	1,287	282
North Cascades	22,291	2,275	370,832	0.064	0.032	0.101	454	118	288

Table A1.2. Spearman's rank correlations (ρ) between annual area burned within each ecoregion and the area of subalpine parkland or alpine vegetation burned (n=29)[a], and results of linear regressions predicting annual area of subalpine parkland or alpine vegetation burned as a function of annual total area (all vegetation types) burned (n = 29). Data were log-transformed prior to analysis.

Ecoregion	Subalpine parkland						Alpine Vegetation					
	ρ	**Intercept**	**Slope**	***t***	***P***	R^2	***ρ***	**Intercept**	**Slope**	***t***	***P***	R^2
Study Area	0.89	-10.67	1.48	7.50	<0.001	0.83	0.88	-11.62	1.45	7.49	<0.001	0.76
Blue Mountains	0.69	-3.69	0.64	4.79	<0.001	0.29	0.69	-2.62	0.47	3.56	<0.001	0.29
Canadian Rockies	0.98	-0.20	0.59	8.77	<0.001	0.88	0.90	-0.24	0.33	4.97	<0.001	0.58
Cascades	0.82	0.01	0.44	6.27	<0.001	0.58	0.92	-0.10	0.33	4.75	<0.001	0.72
Columbia Mountains [b]	0.65	-0.71	0.28	3.17	0.002	0.29	…	…	…	…	…	…
Eastern Cascades [b]	0.63	-0.58	0.19	1.86	0.063	0.15	…	…	…	…	…	…
Idaho Batholith	0.92	-3.70	0.85	6.98	<0.001	0.68	0.76	-3.30	0.54	4.50	<0.001	0.35
Middle Rockies	0.88	-2.80	0.64	5.93	<0.001	0.48	0.87	-3.51	0.79	7.40	<0.001	0.46
North Cascades	0.84	-0.58	0.65	8.38	<0.001	0.63	0.78	-0.63	0.28	3.68	<0.001	0.29

[a] All correlations were significant ($P < 0.001$).

[b] Alpine vegetation classes in the Columbia Mountains and Eastern Cascades Ecoregions were not evaluated because there was little alpine vegetation in those ecoregions.

Table A1.3. Wilcoxon Signed Rank Test comparing the annual proportion of subalpine parkland or alpine vegetation burned and the annual proportion all vegetation types burned. Significant results with negative median support the hypothesis that subalpine parkland or alpine vegetation is less likely to burn than the broader landscape. (There were no tests with a significant result and a positive median suggesting greater likelihood of burning than the broader landscape).

Ecoregion	Subalpine Parkland			Alpine Vegetation		
	V	***P***	**Estimated Median**	***V***	***P***	**Estimated Median**
Study area	141	0.100	-0.0004	29	0.000	-0.0017
Blue Mountains	86	0.008	-0.0014	147	0.206	-0.0009
Canadian Rockies	105	0.185	0.0005	13	0.003	-0.0011
Cascades	61	0.737	-0.0003	23	0.021	-0.0007
Columbia Mountains [a]	81	0.050	-0.0002	…	…	…
Eastern Cascades [a]	55	0.002	-0.0013	…	…	…
Idaho Batholith	78	0.005	-0.0014	0	0.000	-0.0059
Middle Rockies	73	0.006	-0.0003	25	0.000	-0.0008
North Cascades	65	0.048	-0.0007	30	0.002	-0.0018

[a] Alpine vegetation classes in the Columbia Mountains and Eastern Cascades Ecoregions were not evaluated due to low cover.

CHAPTER 2: FIRE INFLUENCES FOREST STRUCTURE IN ALPINE TREELINE ECOTONES

Abstract

I studied the effects of fire on vegetation structure within four alpine treeline ecotones extending from *Abies lasiocarpa/Picea engelmannii* forests at lower elevations through *Pinus albicaulis*/*Larix lyallii* parkland to alpine tundra at higher elevations. Two sites were in the Northern Cascade Range and two were in the Northern Rocky Mountains; all had burned 18-27 years prior to sampling. I assessed the likelihood of different pre-fire canopy-cover structural classes—closed forest (>40% tree cover), open forest (10%-40%), parkland (<10%) and non-forest areas—to burn and to change to a different structural class after fire. I also evaluated changes in forest structure—specifically the abundance of live trees within five diameter at breast height (DBH) classes—and tree species composition, using non-metric multidimensional scaling (NMDS) to visualize differences and Permutational Multivariate Analysis of Variance (PERMANOVA) to test statistically for differences from pre-fire to post-fire, and between unburned and three higher-severity classes.

Non-forested areas were less likely to burn than the landscape as a whole, and across each ecotone, fire increased the proportion of non-forested area. Fire severity and the effects of the fire on forest structure were mixed: previously forested stands had a greater probability of retaining forest cover than they had of becoming non-forested and dense close-forest stands were never converted to non-forest, even when they burned with high severity. Greater fire severity decreased the abundance of larger, relative to smaller, overstory trees; the latter suffered greater mortality. Of the four common high-elevation tree species observed in burned plots, *Abies lasiocarpa* had the highest rates of mortality (60%), *Larix lyallii* had the lowest rate (11%), with intermediate levels in *Pinus albicaulis* (52%) and *Picea engelmannii* (37%). Higher severity fire

was associated with a significant change in species composition, which correlated most strongly, in ordination space, to decreases in the relative abundance of *A. lasiocarpa* and *P. engelmannii*. Burned plots that had been open forest or parkland before the fire but were now non-forest had post-fire species composition that were intermediate between unburned forest plots and unforested plots, indicating that their ground-level species composition had shifted to become more similar to the non-forest plots.

Introduction

The influences of climate and local feedbacks on tree establishment and growth in the alpine tree-line ecotone (ATE)—the transitional area comprising the upper edge of closed forest to treeless alpine—have been studied extensively over the past century (Körner 2012). Multiple lines of evidence indicate that upright tree growth in the ATE is strongly related to climate. These include (1) observational studies of vegetation distribution (Daubenmire 1954), (2) the relationships between climate and treeline fluctuations during the Holocene (Rochefort et al. 1994), (3) correlations between tree growth (tree ring width) and climatic variables, and (4) the strong influence of microtopography and neighboring vegetation on tree establishment and persistence (Callaway et al. 2002, Malanson et al. 2011). Increased tree establishment in the ATE and an upward movement of treeline is expected under future warmer climate (Malanson et al. 2011). Recent observations suggest that these changes are already occurring in some ecosystems (Danby and Hik 2007, Harsch et al. 2009).

In western North America, climate warming has also contributed to the increasing area burned by wildfires (Littell et al. 2009, Wotton et al. 2010) and in some places to increased severity and size of high-severity patches within these fires (Cansler and McKenzie 2014). Further increases in area burned are expected under climate change (Flannigan et al. 2009, Littell et al. 2010). The conditions that are likely to enhance tree establishment in the ATE—decreased snowpack and longer growing seasons—are also likely to increase the prevalence of wildfire.

Because regeneration and tree establishment proceed slowly in the ATE, disturbance history and past climate can have persistent influences on pattern and process. These may encompass anthropogenic disturbances such as firewood gathering, grazing, and intentional burning (creating “anthropogenic treelines” as described by Holtmeier and Broll 2005); natural disturbances, such as insects and fire; and severe climate events. All of these shaped past vegetation but may no longer be evident (Malanson et al. 2011). For example, grazing, mining, and anthropogenic burning have maintained treeline in the European Alps at an average of 150-300 meters below climatic limits (Malanson et al. 2011). Similarly, fires of either natural or anthropogenic origin likely limited the upward expansion of tropical treeline through much of the Holocene (Di Pasquale et al. 2008). In fact, in some regions, such as the European Alps, disturbance history may be the primary determinant of current treeline (Holtmeier and Broll 2007). Despite an abundance of research on climatic controls of tree distributions in the ATE and the acknowledged importance of site history in the ATE of Europe, relatively little research has addressed the potential influence of disturbance on the ATE of western North America (Whitesides and Butler 2010, Malanson et al. 2011).

Research conducted in western North America, though limited, suggests that even if wildfires are small in the ATE, their effects can be more persistent than those at lower elevations. On the western side of the Cascade Range in Washington, tree regeneration did not peak until 30-50 years after wildfire (Agee and Smith 1984, Little et al. 1994) or was minimal even 29 years after fire (Douglas and Ballard 1971). In continental climates regeneration appears to peak earlier (*e.g.*, 17-25 years; Tomback et al. 1993), perhaps because the growing season is longer, or early-seral (open-site) pine species are abundant. However, in the drier central Rocky Mountains, abiotic stress (Billings 1969) or competition from herbaceous species (Stahelin 1943) can greatly limit post-fire regeneration, effecting a switch in ecosystem state to dominance by forbs or graminoids.

It is unclear how the distribution and structure of vegetation will change in the ATE in response to the direct and indirect effects of a warming climate and associated changes in fire regimes (Figure 2.1). Recent increases in area burned in the ATE (Chapter 1) may maintain or increase the area of non-forested vegetation, or, alternatively, could facilitate post-disturbance tree regeneration, and thus may change the location and form of the ecotone boundary. Increasing fire may also change community structure and composition within the ATE, *e.g.*, increasing the prevalence of early-successional tree, shrub, and herbaceous species. Because changes in vegetation structure and distribution can have large and lasting effects on ecosystem functions such as wildlife habitat, hydrologic and nutrient cycling, and carbon sequestration, research on the effects of fire in the ATE is critical.

In this study, I examine variability in fire severity and changes in vegetation structure 18 to 27 years after fires of mixed-severity within four ATE sites in the western U.S. Sampling was designed to capture variability in vegetation structure that characterizes the ATE, as well as the variability in fire effects in these environments. My goal was to elucidate how fire changes the distribution of forested and non-forested vegetation within the ATE, as well as the structure and composition of the vegetation within these types. I address four specific research questions:

1. *(a) What is the probability of burning at different severities by wildfire for the four principal structural classes in the ATE—closed forest, open forest, parkland, and non-forest vegetation (alpine and meadow)? and (b) If structural classes burn, how likely is each to change to a different class?*
2. *How does fire change the size structure and composition of overstory trees and how are these changes mediated by pre-fire conditions?*
3. *(a) How does the composition of ground-layer species vary with fire severity? (b) Do non-forested areas created by fire and non-forest areas that were present before the most recent fire have similar species composition?*

Methods

Study locations

I sampled four high-elevation sites between July and September 2012 on National Forests in the Pacific Northwest and Northern Rockies, USA (Table 2.1, Figure 2.2). All had burned 18-27 years earlier. Sites were bounded at lower elevations by *Abies lasiocarpa/Picea engelmannii* forests and extended upward in elevation through a mosaic of *Pinus albicaulis/Larix lyallii* parklands, and alpine tundra (see Table 2.1 for the dominant tree species at each site). Species nomenclature follows the PLANTS National Database (http://plants.usda.gov). Two fires were in the northern Cascade Range, Hubbard Creek (1985) and Butte Creek (1994); two were in the northern Rockies, the Upper Bear fire (1998) in the Bitterroot Mountains on the Idaho-Montana border, and the Helen Creek fire (1994), in the Bob Marshall Wilderness, Montana.

Sampling methods

Structure plots and gridded plots.—Two types of plots were used: 0.01 ha "structure plots" (20 x 5 m) used to describe stand structure, and "gridded plots" (15 m radius circular plot with forest data subsampled in a 2.82 m radius circle; Table 2.2) used to describe non-forest vegetation. The structure plots were established using a stratified random design where they were stratified within (1) burned and unburned areas and (2) vegetation cover (forest, woodland, and alpine). Burned and unburned areas were identified based on fire perimeters from classified MTBS severity images (Monitoring Trends in Burn Severity; Eidenshink et al. 2007), and vegetation cover was identified from LANDFIRE landcover data (Rollins and Frame 2006). These plots were used to measure change from pre-fire to post-fire, and differences between burned and unburned areas. I found that the LANDFIRE vegetation cover classes and the MTBS severity classes often did not correspond with our observations in the field, so I reassigned plots to a canopy-cover class and burn-severity class (described in detail below) in the field.

The gridded plot locations were used to capture changes in burn severity and changes from fire in tree cover across each study site. In other words, the gridded plots sampled the "ATE site" within and near (within 300 m) the fire, on a systematic grid of points at 150-meter spacing grid using a random start established without bias. The systematic grid allowed for efficient sampling across study areas, and helped capture the fine-scale variability in species composition, structure, and fire effects within the ATE. Gridded or structure plots were not established in areas with permanent snowfields, unvegetated talus fields, evidence of vegetation disturbance due to avalanches, standing water, slopes >100%, or inadequate site safety and accessibility.

Quantifying burn severity and canopy-cover structural classes.—Each plot was assigned to a burn-severity class based on evidence of tree mortality and soil charcoal (Table 2.3; Figure 2.3); classes generally followed burn-severity class descriptions used in other assessments of burn severity in the field (Miller and Thode 2007, Cansler and McKenzie 2012). Plots were also assigned to a "canopy-cover structural class" by visually estimating the percentage canopy cover of trees >1.4 m in height, at the scale of the structure plot, or a 15 m radius circle from the center of gridded plots (Table 2.3, Figure 2.4).

Measuring current forest structure.—Current live overstory trees were sampled within gridded plots and structure plots. The species and diameters at breast height (1.4 m; DBH) of all overstory trees (≥1.4 m height) were measured to the nearest 0.1 m. Based on counts of branch whorls and tree growth morphology made in the field, all overstory trees established before the fire.

Reconstructing pre-fire structure.—Wood decomposition occurs slowly in the ATE, allowing us to reconstruct tree status (live or dead) before the fire. In addition to recording all live trees, I recorded all dead trees ≥1.4 m height that had been standing before the fire. Dead trees were considered to be within the plot if the center of their original rooting location was within the plot boundary. For each dead tree, I recorded whether it was (a) recent snag, with fine branches or brown needles present, (b) dead with evidence of fire but no evidence fire burned into the heartwood, or (c) dead with evidence of fire and evidence the fire burned into the heartwood (Figure 2.5). Using these classes, I reconstructed pre-fire live tree density: if the heartwood of a tree was charred by the fire, it was considered dead at the time of the fire, and not included in the pre-fire live tree pool.

The reconstructed pre-fire structure did not account for changes in the size of trees from pre-fire to the time of measurement. Although tree growth is slow in the ATE, some diameter growth had probably occurred in most trees since the fire. Overall, the uncertainty associated with changes in tree size should be minimal compared to changes in size structure due to fire-caused mortality, since tree sizes were analyzed within size classes (described below).

In most cases (92.5%), I was also able to use bark and branch morphology to determine the species of dead trees. Prior to analyses, the 218 unidentified dead trees (mean = 0.41; *s.d.* = 1.56) that were alive before the fire were assigned to one of three species in the pre-fire data: *A. lasiocarpa, Picea engelmannii,* or *Pinus contorta* (dead trees of all other species could be identified in the field). Assignments were based on the proportional abundance of live trees representing each combination of species x canopy cover class x site. Species density and basal area before and after assigning species to unknown snags were highly correlated (Pearson's *r* ranged from 0.921 to 1.00 for density, and 0.934-1.00 for basal area; Appendix 2.1 Table A3).

<u>Ground-layer species</u>.—In structure plots, cover of all herbaceous and shrub species was estimated using the point-intercept method (Elizinga et al. 1998) with 20 points spaced 0.5 m apart along the long (20 m) axis of the plot. Species were assigned a growth form following Körner (2003) based on observations of growth form in the field (Figure 2.6) and descriptions from the PLANTS National Database (http://plants.usda.gov/).

Full field protocols and data sheets are included in Appendix 2.2.

Statistical analysis

<u>Relationship between canopy-cover classes and other measures of overstory structure</u>.—As a precursor to my primary questions, I tested whether classes (based on field estimates of cover) also differed in other measures of structure (total tree density and basal area across all species),

to aid the interpretation of changes between classes. I used a mixed-effect model for each response variable (density and basal area); site and plot type were treated as random effects and canopy-cover class as a fixed effect. I did not include the non-forested class in this analysis because it lacked trees. This analysis included the pre-fire plot conditions (pre-fire canopy cover class as a predictor of pre-fire density and pre-fire basal area) and post-fire plot conditions (post-fire canopy cover classes as a predictor of post-fire density and post-fire basal area). Prior to analysis, density was square-root transformed and basal area was cube-root transformed; nevertheless, there still were departures from normality. Thus, I conducted a post-hoc comparison to confirm results using the non-parametric the Kruskal-Wallis H test (Zar 2010), to test for differences between the four groups, and the Wilcoxon rank sum test (Zar 2010) to test for differences between adjacent classes (*i.e.*, unforested and parkland, parkland and open forest, open forest and closed forest). Both the parametric and non-parametric tests yielded significant differences in density and basal area between cover classes (Appendix 2.1); I used the latter in subsequent analyses.

Probability of burning at different severities.—To assess how probability of burning relates to initial forest structure (question 1a), I calculated the proportion of gridded plots within each cover class that burned at each level of severity. The closed forest structural class in the Upper Bear and Helen Creek sites was excluded from analyses, and the transitional analysis below, as was the non-forest in Butte Creek, because there were few gridded plots in those classes at those sites. Probabilities were based on the distribution of fire severity among gridded plots, which systematically sampled the ATE site, including burned and unburned areas, not on the structure plots, whose locations were stratified based on forest cover and presence and absence of fire. For each cover class I compared the probability of escaping fire or of burning at low, moderate, or high severity using Pearson's χ^2 tests (Zar 2010). I used contingency tables for this analysis, where the expected number of plots at each severity class for each pre-fire canopy cover class is based on the distribution of plots in each severity class across all canopy cover classes, and the relative abundance of each canopy cover class. *P*-values were computed using a Monte Carlo simulation (Hope 1968, Patefield 1981). This statistical test and all other tests were conducted at $\alpha = 0.05$.

<u>Probability of transition between structural classes</u>.—I assessed the probabilities of changing from one structural class to another due to fire (question 1b). For all gridded plots, and among the subset of burned gridded plots, I calculated transition probabilities between cover classes and tested for differences in pre- and post-fire distributions using a Mantel-Haenszel χ^2 test (Mantel and Haenszel 1959, Mantel 1963). This type of contingency analysis accommodates an additional grouping variable (in this case, site) as a third dimension.

<u>Effects of fire on the size structure and composition of overstory trees</u>.—To assess changes in the structure and composition of overstory trees due to fire (question 2), I used multivariate data visualization and tests of multivariate responses between groups. I use the structure plots for these analyses, since these plots were stratified to capture variation in canopy cover and burned vs. unburned areas within each site, and tree measurements were taken over a larger area, encompassing greater variation in overstory tree structure and composition.

I used several approaches to explore the effects of fire on the structure and composition of overstory trees between cover classes. Forest structure was characterized by the densities of trees of varying size, *i.e.*, as the number of trees in each of five diameter classes. Diameter classes varied in width: A <5 cm; B ≥5 cm to <10 cm; C ≥10 cm to <20 cm; D ≥20 cm to <40 cm; and E ≥40 cm. I used non-metric multidimensional scaling (NMDS; Kruskal 1964a, 1964b, Clarke

1993, McCune and Grace 2002) to illustrate variation in size structure of trees among plots representing initial cover classes and how size structure was changed by fire. The sample matrix comprised both pre- (reconstructed) and post-fire densities of trees in each diameter class in each plot. Density data were square-root transformed before analysis and zero-adjusted Bray-Curtis was used as the dissimilarity measure (Bray and Curtis 1957, Clarke et al. 2006), facilitating inclusion of samples without "species" (in this case, trees size classes). Bray-Curtis has a constant maximum for sites that have no elements in common, and is an optimal compromise between quantitative measures (*e.g.*, Euclidean distance) that may not be robust to zero inflation and those based on presence-absence (*e.g.*, Sorensen's distance) (Clarke et al. 2006, Legendre and Legendre 2012).

To interpret the tree size structure ordination, I calculated the Pearson's correlation coefficient between tree density in each diameter class, and the density of all trees, with the score along each ordination axis. To visualize structural changes in response to fire, I plotted vectors connecting the centroids of samples representing the four cover classes before and after fire. To visualize the effects of burn severity I show movement in ordination space by connecting the centroids of samples within each burn severity class from pre-fire to post-fire.

I expected that tree species within the ATE would vary in their probability of burning and tolerance to fire. To understand how the response to fire varied by species, I examined changes in tree species composition directly, by calculating the percentage mortality of each species, within each site and pre-fire structure class. I also plotted pre- and post-fire tree density of each species, for each burn-severity class to visualize whether different species burned with different severities.

To illustrate effects of fire on species composition, I performed a similar ordination of the plot compositional data (density of trees of each species). Before conducting the ordination of tree species data, I removed one species, *T. mertensiana*, because it occurred on only two plots. I relativized tree species data by species maximum and then by plot total, equalizing the emphasis on species and plots (Bray and Curtis 1957, McCune and Grace 2002). Therefore, this is an evaluation of the change in the relative abundance of tree species. Data were square-root transformed before analysis, and the zero-adjusted Bray-Curtis distance was used to calculate dissimilarities (Bray and Curtis 1957, Clarke et al. 2006). Like the tree structure ordination, the tree species ordination ordinations was based on 400 iterations with up to 40 random starting configurations; random starting configurations were tested until two similar solutions with minimum stress are found (McCune and Grace 2002, Oksanen et al. 2015). I chose the final number of dimensions by comparing ordinations with 1-6 dimensions based on a scree plot (stress vs. dimensionality; McCune and Grace 2002). The final solution was rotated orthogonally to its principal components to maximize variation along the axes, and centered along axes. NMDS was run in the Vegan package using the metaMDS (Oksanen et al. 2015) in the statistical program R (R Core Team 2014).

I also tested for differences in the multivariate response of (1) overstory tree density within five tree size classes and (2) overstory tree relative density by species between the following groups (predictor variables or treatments):

(1) pre-fire to post-fire across all plots
(2) pre-fire to post-fire in only plots that burned
(3) post-fire, between burned and unburned plots
(4) post-fire, between unburned plots and:
    a. low severity,
    b. moderate severity
    c. high severity

Comparisons between unburned plots and the three levels of severity were run in one combined model, with contrasts limited to the three comparisons of unburned to the higher-severity classes. I ran each of the four models above for all structure plots, and individually for structure plots in each of the four sites: this means there are 20 models per response variable (tree size classes and tree species), and 40 models total.

Differences in the multivariate response were tested using a Permutational Multivariate Analysis of Variance (PERMANOVA) (Anderson 2001, Anderson and Ter Braak 2003). PERMANOVA is a non-parametric test that uses a permutation test instead of an F-test to compare either univariate or multivariate responses to two or more treatments. It uses a distance metric, instead of sums of squares, to measure variation between sample units and group means. When significant differences between groups are detected with PERMANOVA, it may reflect differences in the group means, or the dispersion of the sample units in multivariate space around the group means, or both (Anderson et al. 2008). The significance of individual contrasts was determined by an analogous permutational procedure.

I made the same data adjustments and used the same distance measurements for the PERMANOVA analysis as I did for the ordination. Tree structure data were square-root transformed before analysis and zero-adjusted Bray-Curtis was used as the dissimilarity measure ((Bray and Curtis 1957, Clarke et al. 2006). For the PERMANOVA analysis of tree species data I relativized tree species data by species maximum and then by plot total, equalizing the emphasis on species and plots, and then used the zero-adjusted Bray-Curtis as the dissimilarity measure (Bray and Curtis 1957, McCune and Grace 2002). The PERMANOVA analyses were conducted using 999 permutations, and implemented in the Vegan package (Oksanen et al. 2015) in the statistical program R (R Core Team 2014).

<u>Effects of fire on ground-layer species</u>.—To understand how species composition of the non-

forest species changed due to fire (question 3a), NMDS (as described above) was used to portray the effects of fire on the relative abundance of herbaceous and shrub species. For this analysis, I used post-fire point-intercept data collected on the structure plots; the frequency that each species was encountered was totaled per plot. To keep changes in the abundance of rare species from skewing the results, I removed species that were present on <5% of plots. Plots that then had no species present after removal of rare species were excluded from the analysis. For this ordination, the data were not transformed, but were relativize by species maxima and plot totals, to make this a test of relative species abundance. For these analyses, I used the standard Bray-Curtis distance. I then overlaid the non-forested areas created by fire and non-forest areas that were present before the most recent fire in ordination space to examine visually if and how species composition in these groups differed.

To interpret how fire may have changed the structure and function of ground-layer species at a site, I used the plant growth forms (described above) to interpret the ordination plots. I summed the cover of all species for each plant growth form, calculated their relative abundances, and used the Bray-Curtis distance to measure distances between plots in relation to the relative abundances of growth-form groups. I used two graphical approaches to interpret the changes in species composition via the representation of plant growth form. First, in addition to the sample ordination, I plotted species' centroids in ordination space coded by plant growth form. Second, I correlated the summed abundance of species in each growth form group with plot scores along each ordination axis and plotted significant correlations as vectors in the ordination space (Oksanen et al. 2015).

I used PERMANOVA to address questions 3a and 3b statistically. I tested for differences in the relative abundance of ground-layer species between the different severity classes (question

3a). To understand whether the species composition in non-forested areas created by fire and non-forest areas that were present before the most recent fire were similar or different from each other I tested whether plots that had changed from forested classes to non-forest after fire differed in species composition from non-forested areas that had not burned (question 3b). This second analysis was limited to two sites (Helen and Butte Creeks) because the other sites did not have a sufficient number of plots changing from forest to non-forest. Permutation tests were stratified within sites. I used the same methods of preparing the data for these PERMANOVA analyses as I did for the ordination: species that were present on <5% of plots were removed, plots that then had no species present after removal of rare species were excluded, the data were relativize by species maxima and plot totals, and the standard Bray-Curtis distance was used to measure dissimilarities.

Results

Probability of burning and transition between structural classes (Question 1)

When data from all sites were combined, the probability of pre-fire structural classes burning at a given severity differed from the expectation based on the proportion of area in each structure class (χ^2=54.5, P<0.001; Table 2.4). This was also the case in the Upper Bear and Helen Creek sites (P=0.003 and P<0.001, respectively), but not in the Hubbard Creek and Butte Creek sites (P= 0.828 and P= 0.136, respectively; Table 2.4). Non-forest was consistently less likely to burn and forest more likely to burn than would be expected given their proportion of cover within the site the site (Figure 2.7). Open forest consistently burned at high severity more often than parkland, but closed forest was most likely to be unburned at the Butte Creek site, and burn with high severity at Hubbard Creek (Table 2.4, Figure 2.7).

The overall distribution of canopy-cover classes changed significantly in response to fire

(Cochran-Mantel-Haenszel M^2 test; $\chi^2 = 180$ on 3 *d.f.*, $P<0.001$). There was a significant increase in the proportion of non-forest and a significant decrease in open and closed forest (Mantel-Haenszel χ^2 test; Table 2.5). There was not a significant change in parkland across all sites, and the direction of change differed between individual sites (Figure 2.8). Change probabilities between pre-fire and post-fire canopy-cover structural class reflected the overall decreases in forest cover, but also the moderate severity of the fires. Non-forest remained non-forest; parkland was about equally likely to become non-forest or remain parkland (48% and 52%, respectively); and open forest was more likely to change to parkland (48%) or non-forest (29%) than to remain in the open-forest class (Table 2.6, Figure 2.8). In contrast, closed forests remained in one of the forested classes. High-severity fire did not always cause a change from forest to non-forest. For example, >60% of parkland plots burned at high severity but only 48% of them were non-forest after fire. Likewise, open forest plots were more likely to burn with high severity in Upper Bear than in Helen Creek, but were less likely to become non-forest.

Effects of fire on the structure and composition of overstory trees

<u>Tree size structure.</u>—Correlations between tree density and plot scores indicate that fire decreased the density of all size classes, but more so in moderate and large sized trees. Figure 2.9a shows that from pre-fire to post-fire plots moved to higher values along Axis 1. Axis 1 had negative correlations with the density of all tree size classes, with the strongest negative correlations in the smaller size classes (Table 2.7). Plots also moved to lower values along Axis 2 indicating decreasing density of the larger trees. Axis 2 had moderate positive correlations with density of the three larger size classes (10-20, 20-40, and ≥40 cm) and negative correlations with density of the two smaller size classes (5-10 and <5 cm). Figure 2.9b shows a similar response in relation to fire severity, with a much greater change in high- and moderate-severity plots than in low-severity plots.

PERMANOVA model results indicated that tree diameter distributions changed significantly from pre-fire to post-fire. Post-fire, there were also significant differences between unburned and burned plots, unburned and low-severity plots, and unburned and high-severity plots when data from all four sites were used in a combined model (for all comparisons P <0.001; Table 2.8). Likewise, models testing for differences in the same groups as above at the individual fire scale were also significant, except when comparing pre-fire and post-fire forest, unburned to low, and unburned to moderate severity at the Hubbard Creek site, and when comparing unburned to low severity at the Butte Creek site. In other words, at those two sites, higher-severity fire drove changes in forest structure, whereas low-severity fire caused little change in tree size distributions.

<u>Tree species composition.</u>—Higher burn severity was associated with increased mortality of all species of overstory trees (Figure 2.10), although plots classified as having higher burn severity had higher tree mortality at the Helen Creek and Butte Creek sites than the Hubbard Creek and Upper Bear sites. Of the four common tree species in the ATE—*A. lasiocarpa*, *L. lyallii* (common in three sites), *P. albicaulis*, and *P. engelmannii*—*A. lasiocarpa* and *P. albicaulis* had more mortality, within sites and across all sites, and *L. Lyallii* had the least mortality (Table 2.9). When calculated based on only plots that burned, and for the moderate- and high-severity classes, *P. albicaulis* had similar or more mortality than *A. lasiocarpa* at all sites other than Hubbard Creek, and more mortality than *P. engelmannii* at all sites. There was no indication that *P. albicaulis* was more likely to survive fire than other species. Tree mortality was highest in the two montane species, *P. contorta and P. menziesii,* both of which were relatively uncommon and found at lower elevations within the study areas.

In the NMDS ordination of overstory tree abundance by species the largest shifts from pre-fire to post-fire (Figure 2.11a) and with increasing burn severity (Figure 2.11b) were along Axis 1, which was strongly negatively correlated with the density of *A. lasiocarpa* and *P. engelmannii*, and weakly negatively correlated with the density of other species (Table 2.10). Axis 2 was positively correlated with the abundance of *L. lyallii*, and negatively associated with *P. contorta*; plots shifted towards higher values along this axis from pre-fire to post-fire, potentially reflecting higher mortality of the montane species (*P. contorta and P. menziesii*) and lower mortality of *L. Lyallii.* Axis 3 seemed to reflect a similar gradient as Axis 2, but reversed: it was positively correlated with the montane species, and negative correlations were with the two high-elevation specialists, *L. lyallii* and *P. albicaulis.* Changes along this axis from pre-fire to post-fire were away from the montane species and to mesic species (*L. Lyallii* and *P. engelmannii)*, but less distinct.

Results of PERMANOVA indicate that burning significantly changed the relative abundance of tree species (Table 2.11). Models comparing pre-fire and post-fire, burned and unburned plots, and unburned plots with plots that burned at higher levels of severity were significant when assessed across all plots, and within the Upper Bear, Helen Creek, and Butte Creek sites. Models for the Hubbard Creek site showed no significant difference between pre-fire and post-fire, but did show significant differences between unburned plots and plots that burned with high severity. This is consistent with the lower burn severity (Table 2.4) and less tree mortality (Figure 2.10) at Hubbard Creek.

Ground-layer species composition (Questions 3)

PERMANOVA results showed that the relative abundance of ground-layer species differed significantly between burned and unburned plots (P<0.001) and between unburned and plots that burned at high severity (P<0.001; Table 2.12), but not between unburned and low and moderate severity (Table 2.12).

The NMDS ordination axes were correlated with the abundance of different plant growth forms: Axis 1 was negatively correlated with dwarf shrubs and shrubs (Table 2.13). Axis 2 was positively correlated with cushion plants and graminoids and negatively correlated with herbaceous perennials. Axis 3 was positively correlated with herbaceous perennials and shrubs and negatively correlated with cushion plants (Table 2.13, Figure 2.12b). In ordination space, both unburned and burned non-forest plots were located with high abundances of graminoids and cushion plants, and a high relative abundance of herbaceous perennials. Unburned closed and open forest plots were associated with dwarf shrubs, shrubs, and succulents.

PERMANOVA results (Table 2.12) and ordination overlays (Figure 2.12a) showed that non-forest plots that had forest cover before the fire had a significantly different ground-layer species composition than non-forest plots. Burned and unburned non-forest plots did not differ at the Helen Creek and Butte Creeks sites. In ordination space, burned plots that had been open forest or parkland before the fire, but were now non-forest, were intermediate between unburned forest plots and unforested plots, indicating that their ground-level species composition had shifted to become more similar to the non-forest plots. This likely reflects a decrease in the relative abundance of shrub and dwarf shrub cover, compared to unburned forested plots, and potentially an increase in the dominance of herbaceous perennials.

Discussion

Impacts of fire on ecotone position, cover types, forest structure, and tree species composition

An overarching concern regarding ATEs in a warming climate is whether treeline will move up in elevation and will infill previously treeless areas, or if fire, other disturbances, and edaphic controls will maintain or depress treelines and maintain or increase non-forest areas. Increased fire size and severity could counter the expected expansion of forests upslope into non-forest areas with climate warming. My analysis reveals patterns in post-fire structure and composition that will inform inferences about directional movement and other changes in the ATE. If fire rarely burned non-forested areas and areas where more recently established trees were present, we would expect treelines to advance due to the direct effects of a warming climate. If fires burned throughout the ATE with high severity, we would expect treelines to retreat. My results show that the former pattern was more prevalent than the latter: non-forested areas and smaller (and presumably younger) trees were less likely to burn severely. Nevertheless, the high variability in fire severity and tree mortality led me to consider a third alternative, that changes in the ATE will be complex and variable across ecoregions and site characteristics. This inference is based on how my results differed from those expected from previous research, and on how spatial patterns of fire severity and structural and compositional changes in vegetation present at different spatial scales and interacting with the landscape template.

I expected that if fire reached a plot in the sampled ATE landscape, it would burn with high severity due to the low fire tolerance of most subalpine tree species. To the contrary, my results show that forested areas within the ATE burned with a range of severities. The prevalence of moderate-severity fire differs from observations of fire in *A. lasiocarpa*-dominated subalpine parkland west of the Cascade Range (Agee and Smith 1984, Agee 1993), where fire is often stand-replacing. It also differs from the high-severity fire that typifies closed-canopy subalpine forests, found just below the ATE; fires in closed subalpine forests are often stand-replacing (Agee 1993, Baker 2009). When these forests are dry enough to burn, they are dry enough to support severe fire behaviour (Bessie and Johnson 1995).

In my sites, high fuel moisture and less connected surface and canopy fuels were likely responsible for the variable fire severity. Likewise, the prevalence of smaller trees, which were more likely to survive fire than larger trees, probably due to higher live fuel moisture, also reduced the overall overstory tree mortality in the ATE. This result is in line with observations in subalpine forest that younger stands burn less severely than older stands (Kulakowski and Veblen 2007). Local moist conditions probably also explain why closed forest, within the two of my ATE sites in which they were common, were less likely to burn than stands with lower canopy cover and tree density. In these sites closed forest occurred in concave topographical locations where moisture accumulated; the higher density of trees shading surface fuels probably further increasing fuel moisture. This means that the local conditions—including the pre-fire tree density, size distributions, and other factors that influence fuel moisture—operate as strong bottom-up controls on fire severity within the ATE. The implication of this is that the impact of an individual fire on an ATE site will be highly dependent on the pre-fire stand conditions in that site, and topographically driven variability in fuel moisture. However, a common feature of fire in ATEs may be mixed-severity fire and highly variable levels of tree mortality.

Conversely, sometimes fire was present in areas where I did not expect it to burn, or caused higher mortality of a species than I expected. For example, *L. lyallii* has been characterized as a "fire avoider" (Leiberg 1899, Agee 1993). While it did have lower mortality rates than other species in my study sites, it did not avoid fire completely, especially in the Butte Creek site where it was present in many plots that burned at low and moderate severity. Likewise, I expected that *P. albicaulis* would be more resistant to fire-caused mortality than other species, due to its upright branching structure and thicker bark (Larson et al. 2009, Campbell et al. 2011). Nevertheless, it suffered comparable mortality to *A. lasiocarpa* (52% and 60% in burned plots, respectively), a less fire-tolerant species. This result is similar to that of a study of prescribed fire in *P. albicaulis* stands in the Northern Rocky Mountains, where both whitebark pine and subalpine fir experienced >40% mortality (Keane and Parsons 2010). These mortality levels for *P. albicaulis* could simply reflect highly-variable fire severities and fire-return intervals that have been found for whitebark pine within sites and across its range (Arno and Hoff 1989, Arno 2001). On the other hand, in my study sites whitebark pine may have experienced higher levels of mortality due to fire than it otherwise would have because many *P. albicaulis* that were killed by fire were co-located with dead *P. albicaulis,* which had been killed by bark beetles (*Dendroctonus ponderosae*), white pine blister rust (*Cronartium ribicola*), or other factors. These dead *P. albicaulis* may have served as a vector for ignition of neighboring live *P. albicaulis*. Both these examples, and my overall results of my tree species analysis, show that fire in the ATE is not strongly influencing tree species composition: fire cause moderate but highly variable levels of mortality of all the tree species that were present. Thus, increased fire in the ATE may not have a strong influence on tree species composition in the future through direct mortality, although differences in post-fire regeneration success, which I did not address in this study, may have longer-term impacts on tree species composition.

Even if climate warming promotes more rapid ATE tree regeneration after fire and there is increased tree establishment in previously unforested areas due to climate change, increased tree mortality due to climate-mediated effects on fire regimes could increase the non-forested areas within ATEs. A short-term (20-100 years) overall increase in non-forest areas within ATEs may be likely if fire regimes change more rapidly in response to climate than tree regeneration and maturation. Nevertheless, the maximum elevation at which upright trees can grow will likely continue to move up due to weakening of temperature limitations on growth (Körner 2012). Krummholz and tree islands that are isolated within non-forest areas may be less likely to burn, due to low flammability of adjacent non-forest areas, but may be most responsive to changes in climate because they are currently nearest to the climatic limits of upright tree growth. Therefore, climate-driven upward movement of treeline will likely not be counteracted directly by an increase in area burned in the ATE. Instead, the grain size of vegetation patches in the ATE may increase and the ecotone itself may widen in both directions, into closed forest and alpine parkland, with the expansion downward driven by increased fire and the expansion upward driven by relaxation of climatic controls on tree establishment.

Impacts of fire on non-forest areas and non-tree vegetation in the ATE

My results support that non-forest areas within the ATE, including those below treeline within a mosaic of forest and non-forest, are less likely to burn than forested areas. Sparse surface fuels and high fuel moisture likely limit the spread of surface fire in non-forested communities. In the field, I observed that many non-forested areas had lacked charcoal, even when adjacent forested areas or tree clumps within them had burned severely. Agee and Smith (1984) report similar patterns of burning in the 1978 Hoh fire in subalpine parkland on the Olympic Peninsula, WA, USA: "it burned through forested areas and tree clumps, skipping over subalpine meadows... [and] meadow edges invaded by small trees." Fire behaviour that is extreme enough to produce fire brands may be important, and perhaps necessary, for fire to spread between closed forest and tree clumps in the ATE.

These results imply that non-forest vegetation may act as a local-scale refuge from fire within ATEs. Many alpine floras contain rare and endemic species (Körner 2003), which are likely better adapted to chronic climatic stresses in the ATE than to infrequent disturbance by fire. The low flammability of non-forest areas may help the species persist even if the association between climate and fire causes increased fire frequencies in nearby forested areas. Moreover, my results show that the plant community composition in burned non-forest are not significantly different than unburned non-forest, implying that fire does not cause dramatic change in non-forest plant communities. However, the effect of fire on non-forest vegetation in ATE is one area where more research is needed. This study was limited by focusing on relatively few sites, comparing only post-fire vegetation composition, and not focusing on specific species.

By causing tree mortality and creating non-forest areas, fire may act as a surrogate for the micro-climatic stresses that maintained non-forest areas in the ATE in the past. My results show that in terms of the ground-level species composition, the non-forest areas created by fire are similar to unburned non-forest areas, but are not perfect surrogates. Therefore managers considering removing trees from meadows by fire or other methods to maintain habitat of species (Raymond et al. 2013) cannot simply assume that fire-created non-forest areas will provide the same ecosystem functions as preexisting non-forest areas; they may provide surrogates in some circumstances, but not in others. Monitoring and additional research to determine the role of fire-created non-forest areas in the ATE, particularly research using pre- and post-fire data, would greatly inform the extent to which managers should be actively using prescribed fire and mechanical treatments in ATE to adapt to climate change. Nevertheless, my results do not support the need for any type of post-fire restoration by managers in burned ATEs. All of the species I observed in plots were native, fire effects were generally moderate, and by creating and maintaining non-forest areas fire has the potential to counteract climate-driven decreases in non-forest areas within ATEs. Thus, allowing fires to burn in ATEs should be considered as one of a number of climate-adaptation strategies available to natural resources managers.

Conclusions

This study does not suggest that the ATE will respond simply (by moving up or down) or homogeneously to warming climate and increased fire. Rather, it provides insight into the influence of fire on the vegetation structure and composition in alpine treeline ecotones of the

Cascade Range and Northern Rocky Mountains. Within the burned ATEs, fire effects were moderate. Open forest and parkland stands were about equally likely to become non-forest or maintain some tree cover, and closed forest stands generally saw a decrease in tree cover, but rarely experienced complete conversion to non-forest. Fire also caused higher mortality in larger tree size classes; smaller trees were more likely to survive the fire. Overall, my results indicate that post-fire succession dynamics in alpine treeline ecotones in the northern Cascade Range and Northern Rocky mountains will be strongly influenced by heterogeneous patterns of fire severity, resulting in heterogeneous survival of overstory trees. More broadly, we can expect the spatial complexity of the ATE to increase, and with other effects of climate change on tree regeneration and growth, the width of the ecotone to increase.

Pervious research in ATEs has primarily focused on the physiological mechanisms by which climate limits tree growth, and how local variation at a site, such as micro-topography of facilitators, mediates environmental stress. Although constraints on growth imposed by cold temperature shape the elevational limits of trees, it is becoming clearer that other factors, including drought, edaphic constraints, climatic history, chronic physical damage, and disturbance, impose important limitations at local and regional scales. The potential impacts of climate change on treeline and alpine ecosystems have broadened the goals of research in ATEs: understanding the physiological mechanisms that limit upright tree growth is only one goal. An additional important goal is to provide information on the potential impacts of climate change in ATEs, monitor ongoing impacts, and provide management-relevant projections that can be used for climate-change adaptation. Increased research emphasis on non-climatic controls, particularly disturbance, such as wildfires, that are likely to increase rapidly with climate change, responds directly to the need to understand the factors that limit treeline, and thus maintain open alpine vegetation, across the landscape.

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Tables

Table 2.1. Burned areas sampled.

Fire name	Year burned	State	Coordinates (lat-long DD)	Elevation range (m)	Dominant tree species
Hubbard Creek	1985	WA	47.72, -113.24	2150-2460	*Abies lasiocarpa, Larix lyallii, Picea engelmannii*
Upper Bear	1988	MT & ID	46.13, -114.51	2050-2390	*Abies lasiocarpa, Pinus albicaulis, Pinus contorta* subsp. *latifolia*
Helen Creek	1994	MT	48.49, -120.51	1850-2400	*Abies lasiocarpa, Pinus albicaulis, Picea engelmannii*
Butte Creek	1994	WA	48.35, -120.55	1850-2255	*Abies lasiocarpa, Pinus albicaulis, Larix lyallii*

Table 2.2. Numbers of plots and the conditions sampled (burned and unburned) at each site. Structure plots and gridded plots captured the variability in burn severity and forest structure and composition in the ATEs within and adjacent to each fire.

Fire	Year	Location	Structure burned	Structure unburned	Gridded burned	Gridded unburned	Total
Hubbard Creek	1985	Northern Cascade Range (WA)	21	23	45	10	99
Upper Bear	1988	Bitterroot Mountains (ID)	24	8	54	6	92
Helen Creek	1994	Northern Cascade Range (WA)	48	28	75	38	189
Butte Creek	1994	Bob Marshall Wilderness (MT)	42	26	55	28	151

Table 2.3. Descriptions of canopy cover classes and burn severity ratings. Canopy cover was assessed at the scale of structure plots or over a 15m radius circle for gridded plots.

Structure class	Description
Non-forest	No overstory tree cover.
Parkland	<10% tree canopy cover. Includes plots with only Krummholz trees (prostrate trees <2 m height.)
Open forest	10%-40% tree canopy cover.
Closed forest	>40% tree canopy cover.
Burn severity	**Description**
Unburned	No evidence of fire.
Low	Surface fire; few if any trees killed or only a small portion of area affected. Charcoal is present in the soil or on down woody debris.
Moderate	Surface fire with occasion consumption of individual trees; 20%-70% trees killed. Charcoal is present in the soil or on down woody debris.
High	Continuous surface fire with torching or fire carried through the crown; 50-100% of trees killed. Charcoal is present in the soil or on down woody debris. Post-fire erosion is often evident.

Table 2.4. Proportion of gridded plots within each canopy cover class (rows) that burned at each severity (columns). χ2 statistics and *P*-values are from Pearson's chi-squared test of the null hypothesis that pre-fire structural classes burned with the same proportional distribution of severities as the landscape as a whole.

All fires — **$\chi 2$ = 54.5** — ***P* <0.001**

Pre-fire structural class	Unburned	Low	Moderate	High
Non-forest	0.71	0.07	0.05	0.17
Parkland	0.22	0.15	0.20	0.43
Open forest	0.15	0.19	0.18	0.48
Closed forest	0.20	0.27	0.20	0.33

Hubbard Creek (1985) — **$\chi 2$ = 3.15** — ***P*= 0.828**

Pre-fire structural class	Unburned	Low	Moderate	High
Non-forest	NA	NA	NA	NA
Parkland	0.25	0.12	0.25	0.38
Open forest	0.15	0.11	0.22	0.52
Closed forest	0.09	0.27	0.18	0.45

Upper Bear (1988) — **$\chi 2$ = 19.9** — ***P*= 0.003**

Pre-fire structural class	Unburned	Low	Moderate	High
Non-forest	0.60	0.20	0.20	0.00
Parkland	0.06	0.17	0.31	0.47
Open forest	0.06	0.11	0.11	0.72
Closed forest	NA	NA	NA	NA

Helen Creek (1994) — **$\chi 2$ = 27.9** — ***P* <0.001**

Pre-fire structural class	Unburned	Low	Moderate	High
Non-forest	0.70	0.06	0.03	0.21
Parkland	0.19	0.14	0.15	0.53
Open forest	0.19	0.19	0.14	0.48
Closed forest	NA	NA	NA	NA

Butte Creek (1994) — **$\chi 2$ = 13.6** — ***P*= 0.136**

Pre-fire structural class	Unburned	Low	Moderate	High
Non-forest	1.00	0.00	0.00	0.00
Parkland	0.37	0.16	0.16	0.31
Open forest	0.18	0.32	0.21	0.29
Closed forest	0.67	0.00	0.33	0.00

Table 2.5. Mantel-Haenszel *$\chi 2$* test of the null hypothesis that there were no differences in the pre- and post-fire distributions of structural classes (based on gridded plots). The odds ratio signifies the direction of change: values <1 indicates an increase in frequency, values >1 indicates a decrease in frequency.

Structure classes	***$\chi 2$***	***d.f.***	***P***	**Odds ratio**	***C.I.***
Non-forest	141	1	<0.001	0.064	0.037-0.11
Parkland	4.09	1	0.043	1.347	1.01-1.79
Open forest	55.4	1	<0.001	4.383	2.93-6.56
Closed forest	13.4	1	<0.001	6.653	2.26-19.6

Table 2.6. Transition probabilities of gridded plots between canopy-cover classes: rows are pre-fire classes, columns are post-fire classes.

All fires

Pre-fire	Post-fire			
	Non-forest	**Parkland**	**Open forest**	**Closed forest**
Non-forest	1.00	0.00	0.00	0.00
Parkland	0.48	0.52	0.00	0.00
Open forest	0.29	0.48	0.24	0.00
Closed forest	0.00	0.50	0.50	0.33

Hubbard Creek (1985)

Pre-fire	Post-fire			
	Non-forest	**Parkland**	**Open forest**	**Closed forest**
Non-forest	NA	NA	NA	NA
Parkland	0.25	0.75	0.00	0.00
Open forest	0.17	0.65	0.17	0.00
Closed forest	0.00	0.50	0.50	0.30

Upper Bear (1988)

Pre-fire	Post-fire			
	Non-forest	**Parkland**	**Open forest**	**Closed forest**
Non-forest	1.00	0.00	0.00	0.00
Parkland	0.29	0.71	0.00	0.00
Open forest	0.24	0.76	0.00	0.00
Closed forest	NA	NA	NA	NA

Helen Creek (1994)

Pre-fire	Post-fire			
	Non-forest	**Parkland**	**Open forest**	**Closed forest**
Non-forest	1.00	0.00	0.00	0.00
Parkland	0.77	0.23	0.00	0.00
Open forest	0.47	0.29	0.24	0.00
Closed forest	NA	NA	NA	NA

Butte Creek (1994)

Pre-fire	Post-fire			
	Non-forest	**Parkland**	**Open forest**	**Closed forest**
Non-forest	NA	NA	NA	NA
Parkland	0.32	0.68	0.00	0.00
Open forest	0.30	0.22	0.48	0.00
Closed forest	0.00	1.00	1.00	0.00

Table 2.7. Pearson's correlation coefficients between plot scores on NMDS axes and density of trees in each diameter class.

Diameter class	Axis 1	Axis 2	Axis 3
<5 cm	-0.64	-0.38	-0.04
≥5 cm to <10 cm	-0.62	-0.12	-0.28
≥10 cm to <20 cm	-0.62	0.29	-0.25
≥20 cm to <40 cm	-0.50	0.50	0.27
≥40 cm	-0.24	0.29	0.26

Table 2.8. PERMANOVA model results testing for differences due to fire or fire severity in the size distribution of trees. Four models were run for (1) all plots, and (2) each of the four sites individually: (1) pre-fire to post-fire, (2) pre-fire top post-fire for only the burned plots, (3) unburned plots and burn plot, and (4) unburned plots and each of the higher severity classes (low, moderate, high). For the latter, contrasts were used to compare the unburned class to each of the higher severity classes. Data were square-root-transformed before analysis, and the zero-adjusted Bray-Curtis distance was used to calculate dissimilarities.

Predictor variable	Subset of data	*d.f.*	*F*	R^2	*P*
Pre-fire vs. Post-fire	All plots	1 and 379	30.8	0.08	0.001
	Hubbard Creek	1 and 81	1.2	0.01	0.304
	Upper Bear	1 and 63	8.8	0.12	0.001
	Helen Creek	1 and 113	22.4	0.17	0.001
	Butte Creek	1 and 119	6.5	0.05	0.002
Pre-fire vs. Post-fire	All burned plots	1 and 247	48.9	0.17	0.001
	Hubbard Creek; Burned plots	1 and 37	2.0	0.05	0.117
	Upper Bear; Burned plots	1 and 47	12.2	0.21	0.001
	Helen Creek; Burned plots	1 and 81	43.3	0.35	0.001
	Butte Creek; Burned plots	1 and 79	9.6	0.11	0.001
Burned vs. Unburned	All plots; post-fire status	1 and 189	46.7	0.20	0.001
	Hubbard Creek; post-fire status	1 and 40	4.7	0.11	0.006
	Upper Bear; post-fire status	1 and 31	10.9	0.27	0.001
	Helen Creek; post-fire status	1 and 56	32.9	0.37	0.001
	Butte Creek; post-fire status	1 and 59	5.7	0.09	0.006
Unburned vs. low severity	All plots; post-fire status	1 and 189	12.4	0.04	0.001
Unburned vs. moderate severity			32.2	0.11	0.001
Unburned vs. high severity			56.5	0.20	0.001
Unburned vs. low severity	Hubbard Creek; post-fire status	1 and 40	2.0	0.04	0.108
Unburned vs. moderate severity			1.9	0.04	0.129
Unburned vs. high severity			5.4	0.12	0.003
Unburned vs. low severity	Upper Bear; post-fire status	1 and 31	4.5	0.09	0.020
Unburned vs. moderate severity			5.3	0.10	0.011
Unburned vs. high severity			14.3	0.27	0.001
Unburned vs. low severity	Helen Creek; post-fire	1 and 56	13.3	0.12	0.001

	status				
Unburned vs. moderate severity			11.5	0.10	0.002
Unburned vs. high severity			31.7	0.29	0.001
Unburned vs. low severity	Butte Creek; post-fire status	1 and 59	1.3	0.02	0.275
Unburned vs. moderate severity			13.0	0.16	0.001
Unburned vs. high severity			9.0	0.11	0.001

Table 2.9. Percentage mortality of each tree species

Fire	Species	All structure plots		Burned structure plots	
		Mean	***S.D.***	**Mean**	***S.D.***
All	*A. lasiocarpa*	41	45	60	43
Hubbard Creek		19	33	38	38
Upper Bear		45	48	64	45
Helen Creek		59	48	78	39
Butte Creek		33	41	46	42
All	*L. lyallii*	5	19	11	27
Hubbard Creek		0	0	0	0
Upper Bear		0	NA	0	NA
Butte Creek		11	27	16	32
All	*P. albicaulis*	27	43	52	48
Hubbard Creek		5	22	14	38
Upper Bear		29	45	38	48
Helen Creek		46	50	81	38
Butte Creek		29	44	53	47
All	*P. contorta*	80	36	80	36
Upper Bear		77	37	77	37
Butte Creek		100	0	100	0
All	*P. engelmannii*	26	42	37	46
Hubbard Creek		5	16	8	20
Upper Bear		0	0	0	0
Helen Creek		45	50	71	45
Butte Creek		35	47	41	49
All	*P. menziesii*	80	45	80	45
Helen Creek		50	71	50	71
Butte Creek		100	0	100	0
All	*T. mertensiana*	0	0	0	NA
Hubbard Creek		0	0	0	NA

Table 2.10. Pearson's correlation coefficients between overstory tree species abundance ordination axes and overstory tree species abundance.

Species	Axis 1	Axis 2	Axis 3
A. lasiocarpa	-0.671	-0.068	0.350
P. albicaulis	-0.304	-0.316	-0.567
P. engelmannii	-0.457	0.286	0.275
P. menziesii	-0.045	-0.313	0.070
L. lyallii	-0.217	0.486	-0.292
P. contorta	-0.051	-0.493	0.174
T. mertensiana	-0.044	0.170	-0.026

Table 2.11. PERMANOVA model results testing for differences in the relative abundance of overstory tree species within the three forested canopy-cover classes. Differences from pre-fire to post-fire and between unburned plots and more severely burned plots were assessed.

Predictor variable	Subset of data	*d.f.*	*F*	*P*
Pre-fire vs. Post-fire	All plots	1 and 379	15.1	0.001
	Hubbard Creek	1 and 81	0.2	0.883
	Upper Bear	1 and 63	6.5	0.001
	Helen Creek	1 and 113	14.7	0.001
	Butte Creek	1 and 119	3.9	0.016
Pre-fire vs. Post-fire	All burned plots	1 and 247	25.0	0.001
	Hubbard Creek; Burned plots	1 and 379	0.5	0.684
	Upper Bear; Burned plots	1 and 47	8.3	0.002
	Helen Creek; Burned plots	1 and 81	31.7	0.001
	Butte Creek; Burned plots	1 and 79	6.1	0.004
Burned vs. Unburned	All plots; post-fire status	1 and 189	27.9	0.001
	Hubbard Creek; post-fire status	1 and 40	3.5	0.028
	Upper Bear; post-fire status	1 and 31	6.0	0.003
	Helen Creek; post-fire status	1 and 56	31.0	0.001
	Butte Creek; post-fire status	1 and 59	7.0	0.001
Unburned vs. low severity	All plots; post-fire status	1 and 189	7.7	0.002
Unburned vs. moderate severity			19.6	0.001
Unburned vs. high severity			35.1	0.001
Unburned vs. low severity	Hubbard Creek; post-fire status	1 and 40	2.0	0.144
Unburned vs. moderate severity			2.4	0.072
Unburned vs. high severity			2.5	0.06
Unburned vs. low severity	Upper Bear; post-fire status	1 and 31	1.8	0.165
Unburned vs. moderate severity			3.5	0.034
Unburned vs. high severity			9.8	0.001
Unburned vs. low severity	Helen Creek; post-fire status	1 and 56	12.4	0.002
Unburned vs. moderate severity			13.0	0.001
Unburned vs. high severity			28.7	0.001
Unburned vs. low severity	Butte Creek; post-fire status	1 and 59	4.4	0.014
Unburned vs. moderate severity			10.7	0.002
Unburned vs. high severity			5.7	0.004

Table 2.12. PERMANOVA model results testing for differences in herbaceous and shrub species composition. All permutations were constrained within site and plot type groups. The Bray-Curtis distance was used to calculate dissimilarities. Species data were relativized by species maxima and site totals.

Predictor variable	Subset of data	*d.f.*	*F*	*P*
Burned vs. Unburned	Structure plots	1 and 215	3.42	<0.001
Unburned vs. low severity	Structure plots	1 and 215	2.13	0.127
Unburned vs. moderate severity	Structure plots	1 and 215	1.55	0.017
Unburned vs. high severity	Structure plots	1 and 215	5.48	<0.001
Pre-fire non-forest unburned vs. non-forest burned	Post-fire non-forest structure plots; Helen Creek and Butte Creek sites	1 and 62	1.30	0.100
Pre-fire non-forest, that was unburned vs. pre-fire parkland that is now non-forest	Post-fire non-forest structure plots; Helen Creek and Butte Creek sites	1 and 62	2.64	<0.001
Pre-fire non-forest, that was unburned vs. pre-fire open forest that is now non-forest	Post-fire non-forest structure plots; Helen Creek and Butte Creek sites	1 and 62	3.39	<0.001

Table 2.13. Pearson's correlation coefficients between ground-level species ordination axes and relative abundance of growth forms of ground-level species.

Growth form	Axis 1	Axis 2	Axis 3
Cushion plant	0.08	0.40	-0.28
Dwarf shrub	-0.42	-0.11	-0.04
Graminoid	0.02	0.35	-0.12
Herbaceous perennial	0.20	-0.38	0.36
Shrub	-0.23	-0.13	0.28
Succulent	-0.18	0.14	0.07

Figures

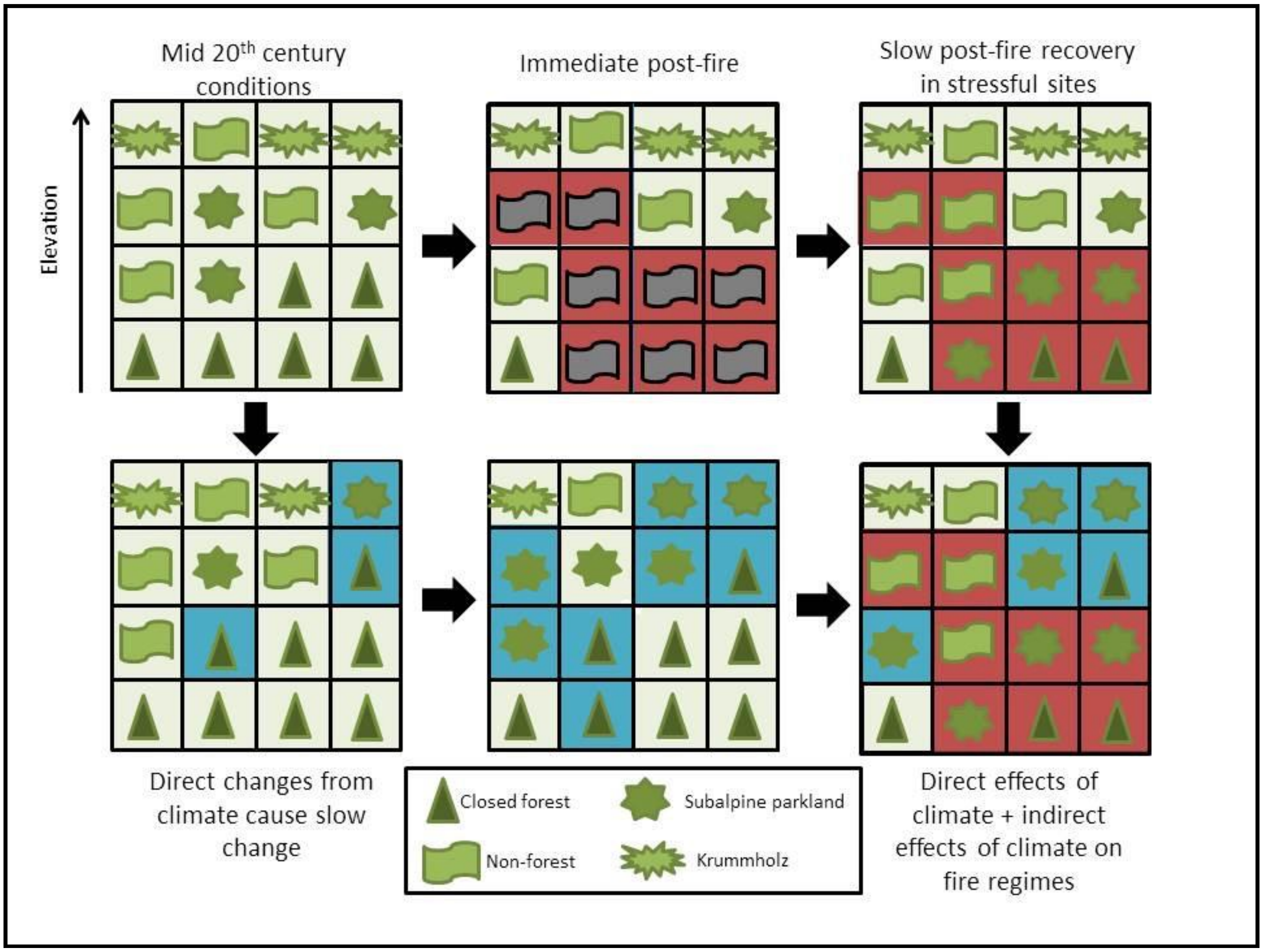


Figure 2.1. Conceptual model of possible state change in the ATE due to fire (top row) and climate change (bottom row).

Each four-by-four grid represents a simplified cover map of the same ATE with increasing elevation along the vertical axis. Changes over time from fire and the direct effects of climate are shown as transitions from one four-by-four grid to the next. The extent of one hypothetical fire is shown (top middle), after which recovery at higher elevations occurs more slowly than recovery at low elevations: probability of a persistent change to a different structural type after disturbance is influenced by the severity of the fire and the position on the landscape (*e.g.*, environmental stress at that location). Likewise, changes in local climate may also cause state changes, as shown in blue cells on the bottom left and middle. The type of change caused by climate change likely differs in direction (increase in tree cover) from those caused by fire (decrease in tree cover), making the combined influence of fire and climate change different from the effect of climate or fire alone (bottom right).

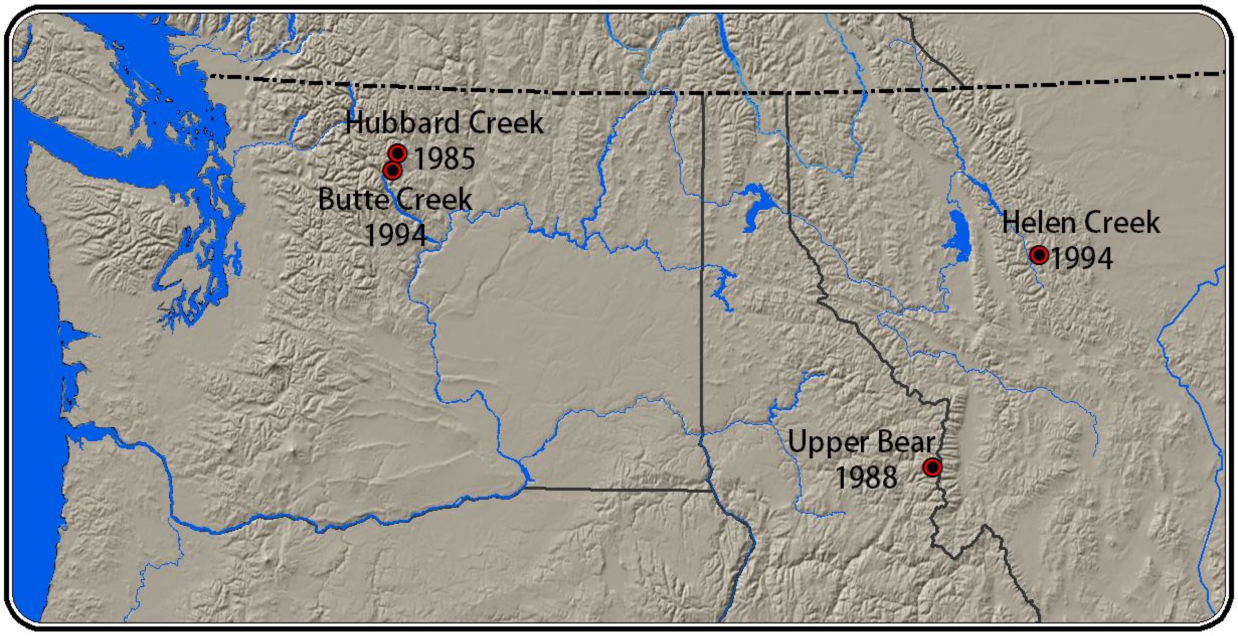


Figure 2.2. Locations of field sites.

Figure 2.3. Examples of burn severity classes. Top left – unburned; top right - low; bottom left – moderate; bottom right - high.

Figure 2.4. Examples of pre-fire canopy-cover classes determined in burned field site. Top left - non-forest; top right - parkland; bottom left - open forest; bottom right - closed forest.

Figure 2.5. Evidence used to assess timing of mortality. Left: Boles of larger trees are deeply charred and were presumed dead before the fire. Smaller trees were presumed alive before the fire since only their bark, fine branches were removed, and their heartwood was not charred: Middle: Bole of log deeply charred, primarily burned on underside: presumed dead and down before the fire. Right: bark and fine branches still present; beetle galleries with frass under bark, presumed to have died recently

.

Figure 2.6. Ground-layer species growth forms. Example of plant species from each growth form group. Top left - cushion plants, *Antennaria lanata*. Top middle - dwarf shrubs: *Phyllodoce empetriformis*. Top right - shrubs, *Menziesia ferruginea.* Bottom left - graminoids: *Festuca viridula*. Bottom middle - herbaceous perennials, *Castilleja miniata*. Bottom right - Succulents, *Sedum divergens.*

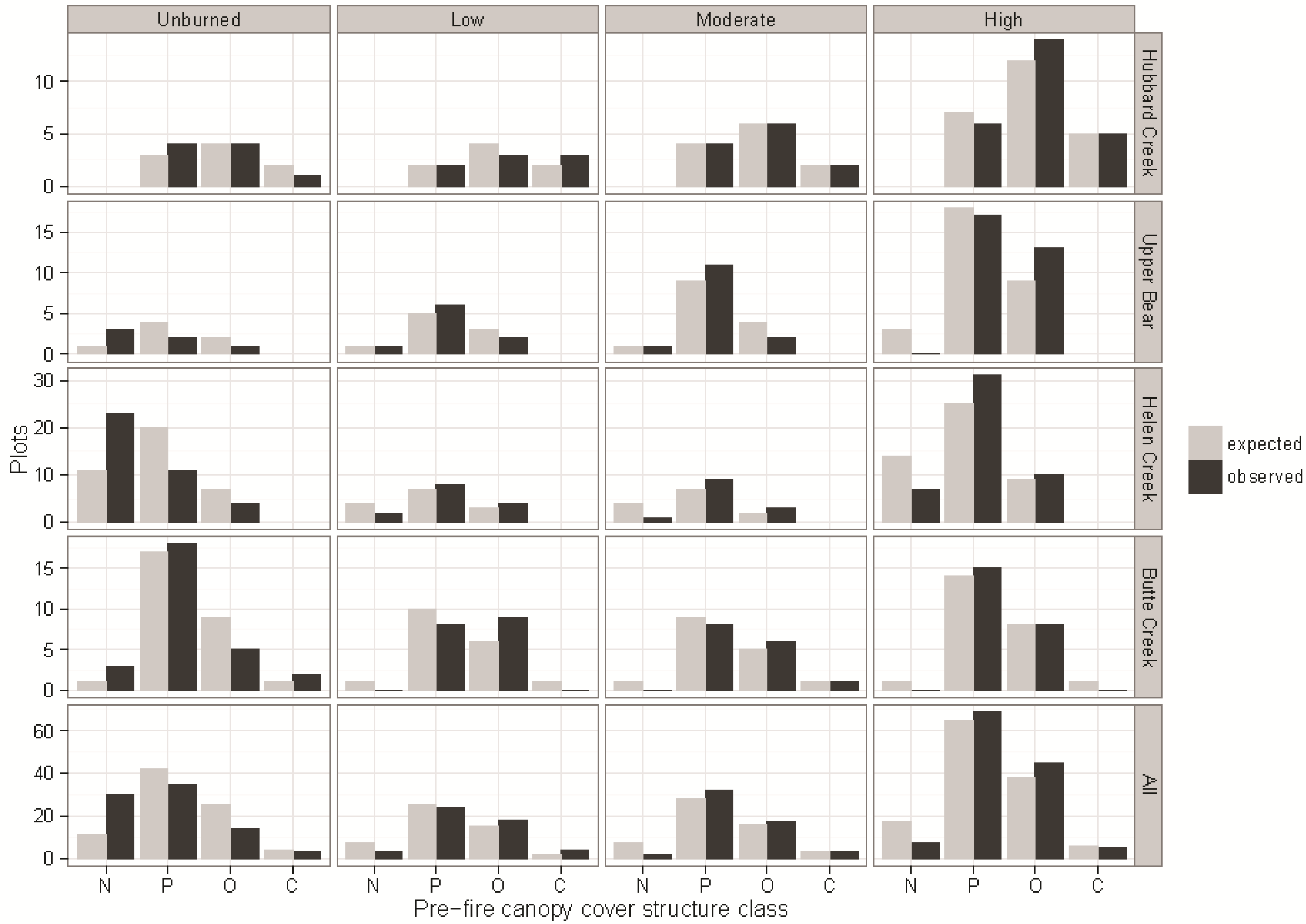


Figure 2.7. Observed and expected plots burned in each canopy-cover class. Observed values (black) are the number of gridded plots burned at a given severity within a cover class; expected numbers of plots (grey) are based on the proportional abundance (cover) of plot in each structural class and the proportion of plots across all structural classes burned at a given severity. X-axis labels are pre-fire canopy-cover classes: N = Non-forest; P = Parkland; O = Open forest; C = Closed forest. Each panel represents a burn severity class x site (or for the bottom row, all sites combined).

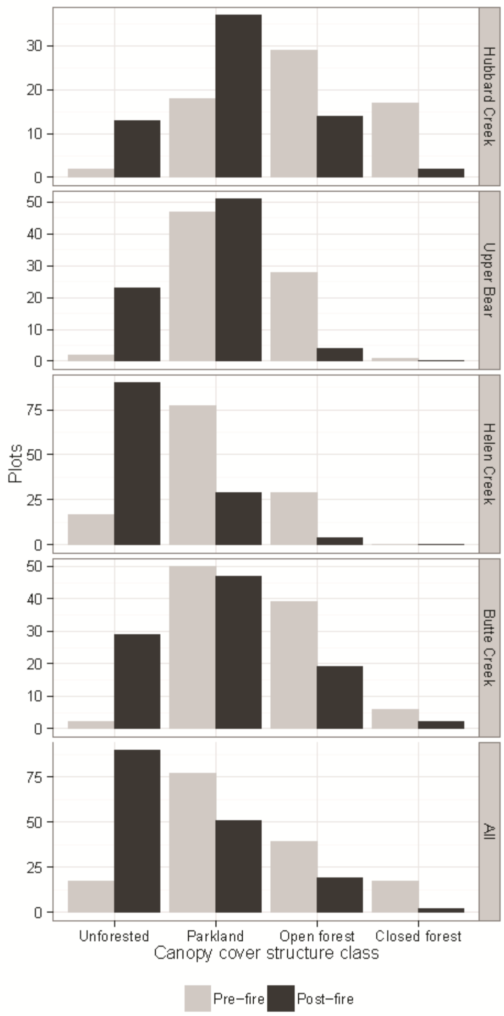


Figure 2.8. Number of plots in each pre- and post-fire cover class.

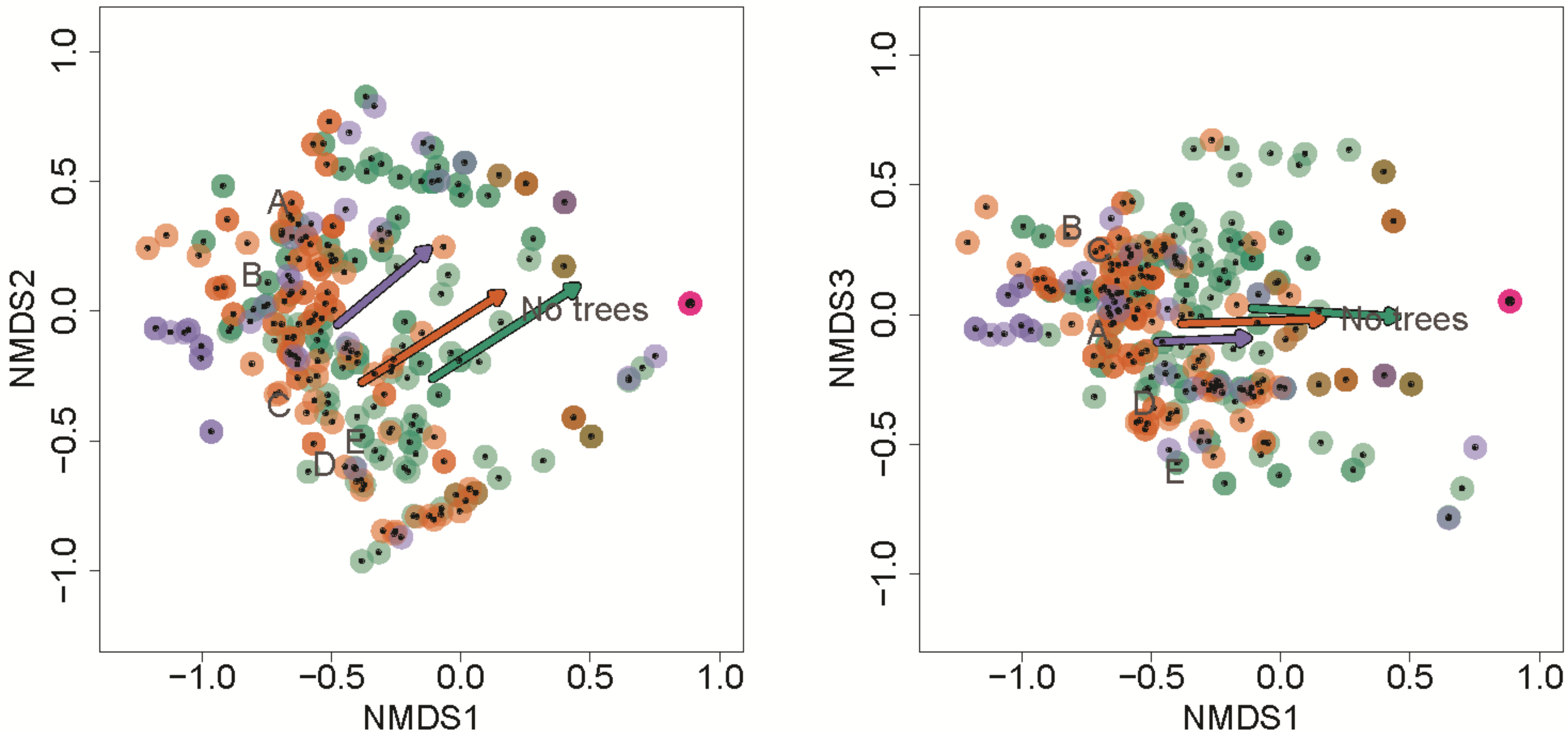


Figure 2.9a. A three-dimensional NMDS ordination of tree density within each of the five tree size classes, using data from all structure plots. The final ordination data had a stress of 8.9, and two convergent solutions were found after 34 tries. Canopy-cover structure classes are plotted in different colors: non-forest – pink (all non-forest plots were located in same place), parkland – teal; open forest – orange, closed forest – purple. Arrows show change from pre-fire to post-fire in plots that burned; colors of arrows match colors of points for each pre-fire canopy-cover structure class. Letters refer to the location of different size-structure classes in ordination space: A <5 cm; B ≥5 cm to <10 cm; C ≥10 cm to <20 cm; D ≥20 cm to <40 cm; and E ≥40 cm.

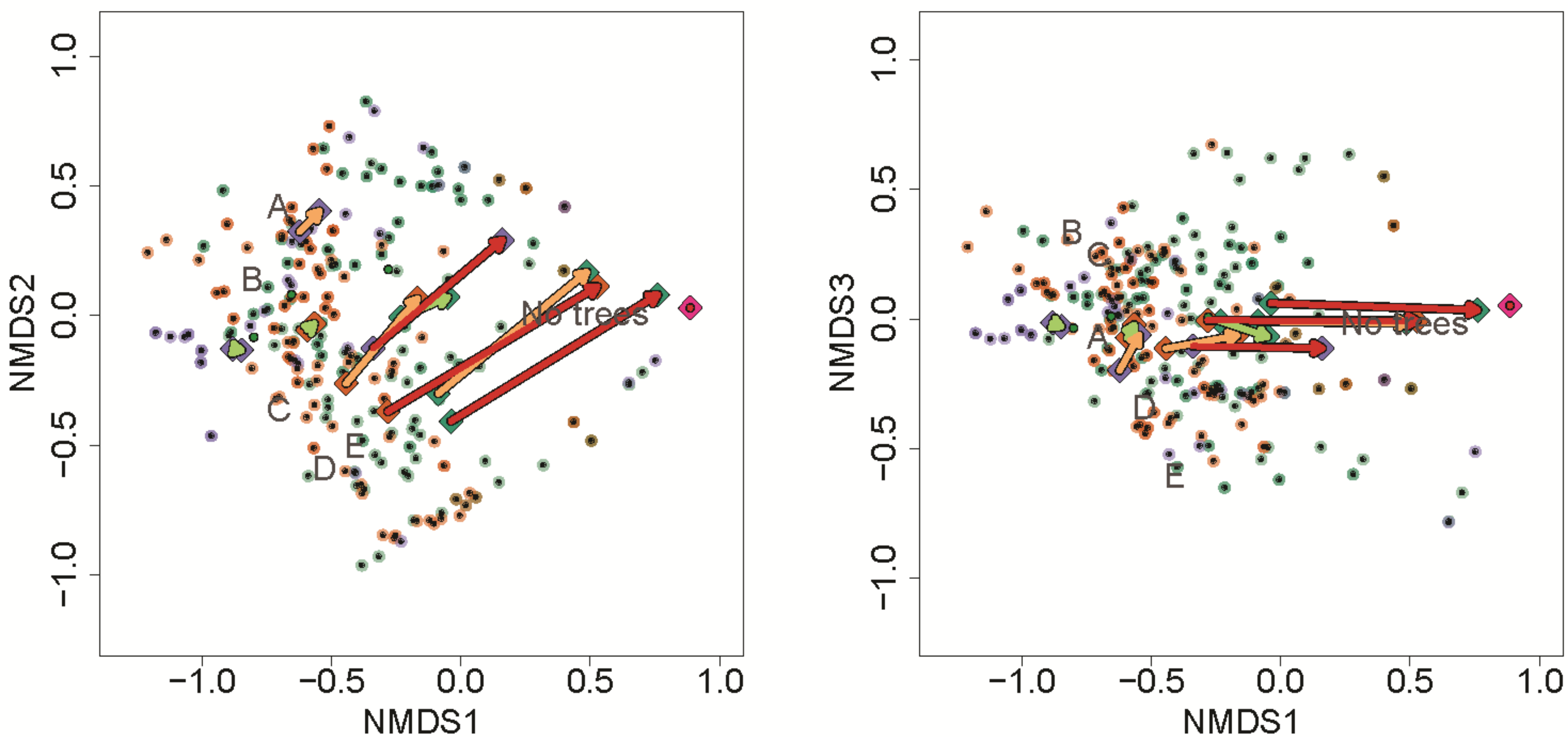


Figure 2.9b. A three-dimensional NMDS ordination of tree density within each of the five tree size classes, showing differences in relation to fire severity. Smaller circular points show locations of plots in ordination space, and are colored by canopy-cover structure class: non-forest – pink (all plots were located in same ordination space), parkland – teal; open forest – orange, closed forest – purple. Arrows connect centroids of each canopy cover class from pre-fire to post-fire, for a given level of burn severity. Arrows are colored by the burn severity class: low severity (green), moderate severity (orange), and high severity (red). The diamond-shaped starting and ending points of the arrows are colored by canopy cover class, using the same colors as for plots. Letters refer to the location of different size-structure classes in ordination space: A <5 cm; B ≥5 cm to <10 cm; C ≥10 cm to <20 cm; D ≥20 cm to <40 cm; and E ≥40 cm.

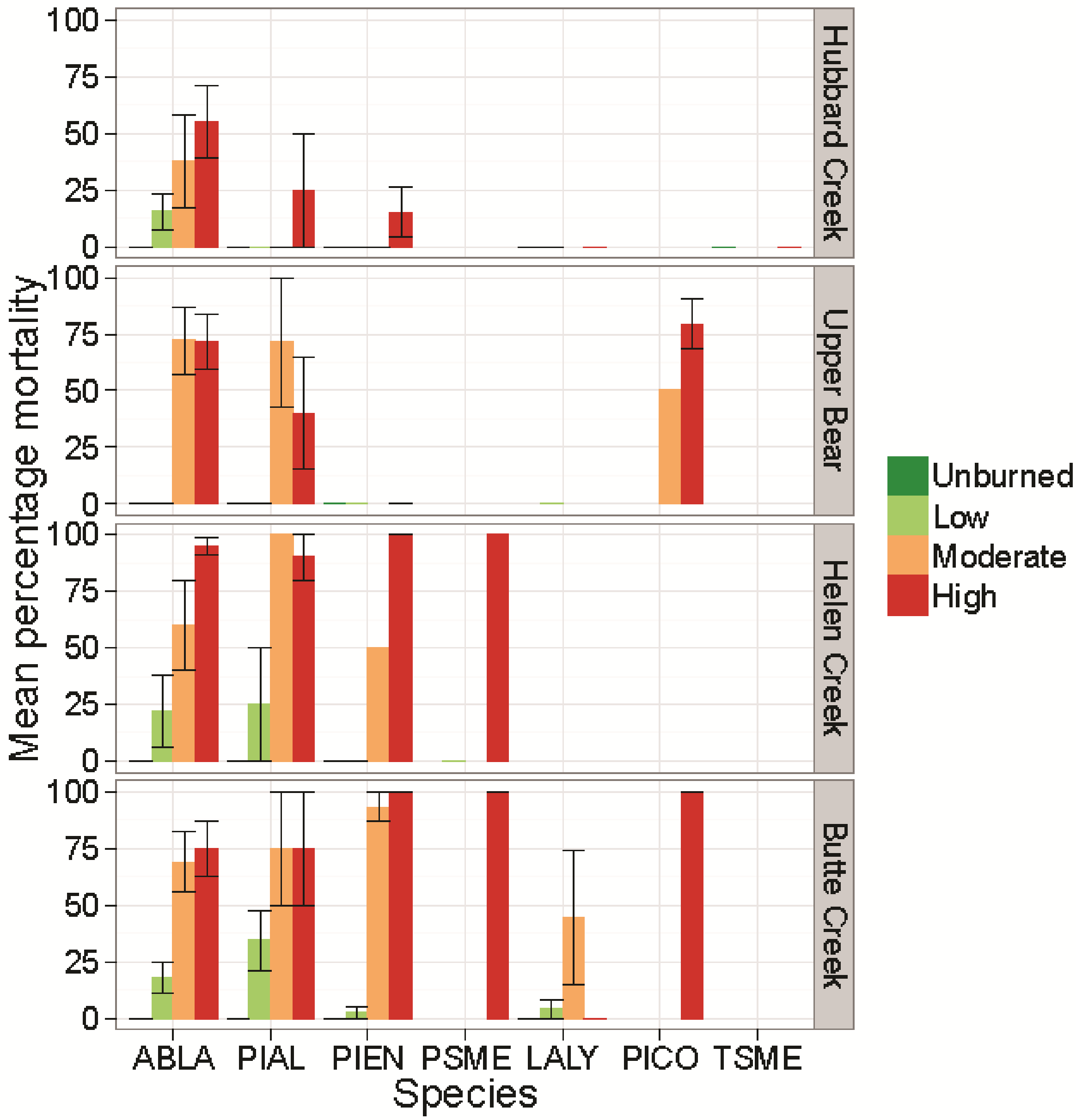


Figure 2.10. Barplots of the mean percentage mortality of each overstory tree species by burn severity class. Error bars show one *S.E.* Species codes are ABLA = *Abies lasiocarpa*; PIAL = *Pinus albicaulis*; PIEN = *Picea engelmannii*; PSME = *Pseudotsuga menziesii*; LALY= *Larix lyallii*; PICO = *Pinus contorta* subsp. *latifolia*; TSME = *Tsuga mertensiana*.

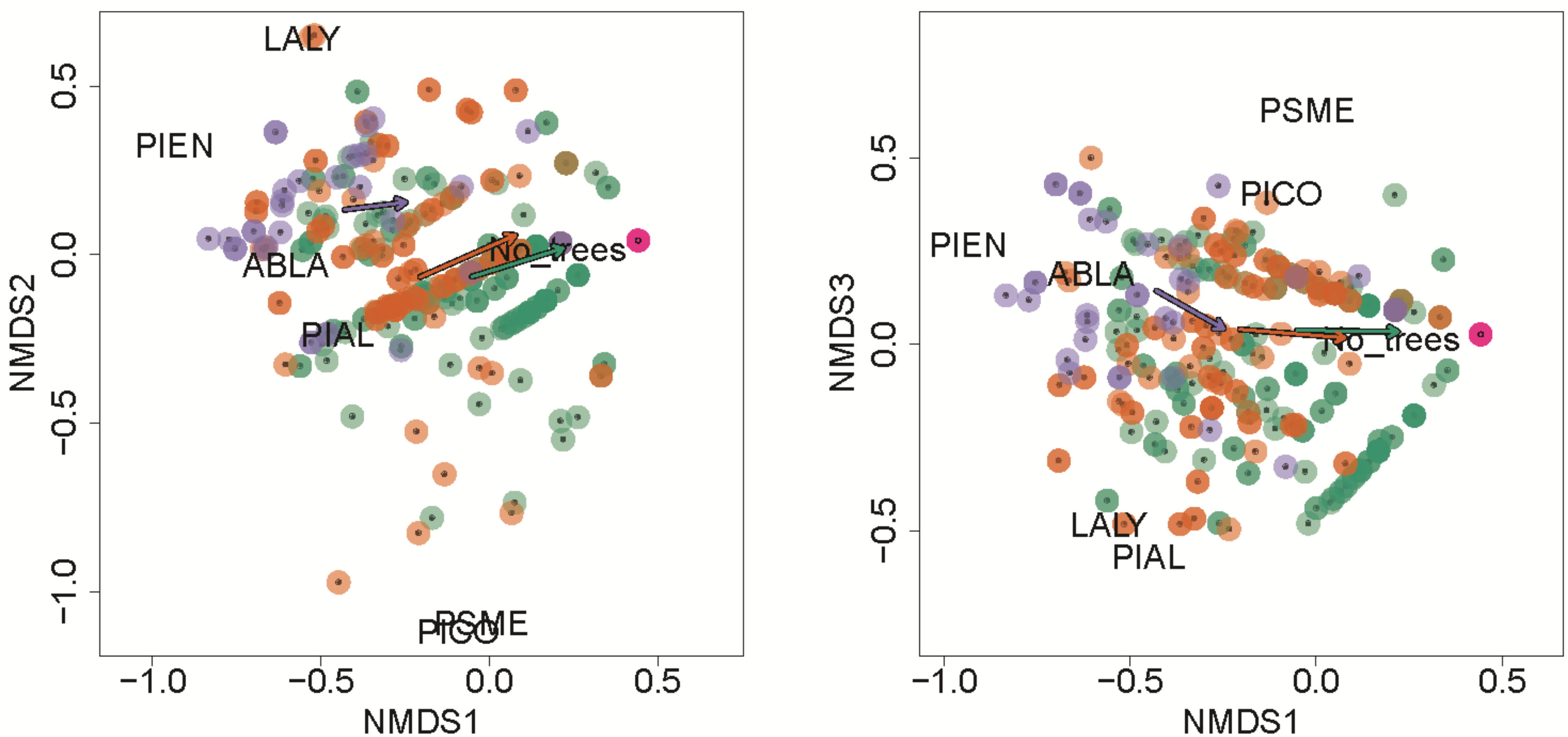


Figure 2.11a. A three-dimensional NMDS ordination of density of tree species, using data from all structure plots. The ordination had a final stress of 8.8 and two convergent solutions were found after 6 tries. Canopy-cover structure classes are plotted in different colors: non-forest – pink (all plots were located in same ordination space), parkland – teal; open forest – orange, closed forest – purple. Arrows show change from pre-fire to post-fire in plots that burned; colors of arrows match colors of points for each pre-fire canopy cover class. Species codes are ABLA = *Abies lasiocarpa*; PIAL = *Pinus albicaulis*; PIEN = *Picea engelmannii*; PSME = *Pseudotsuga menziesii*; LALY= *Larix lyallii*; PICO = *Pinus contorta* subsp. *latifolia*; TSME = *Tsuga mertensiana*.

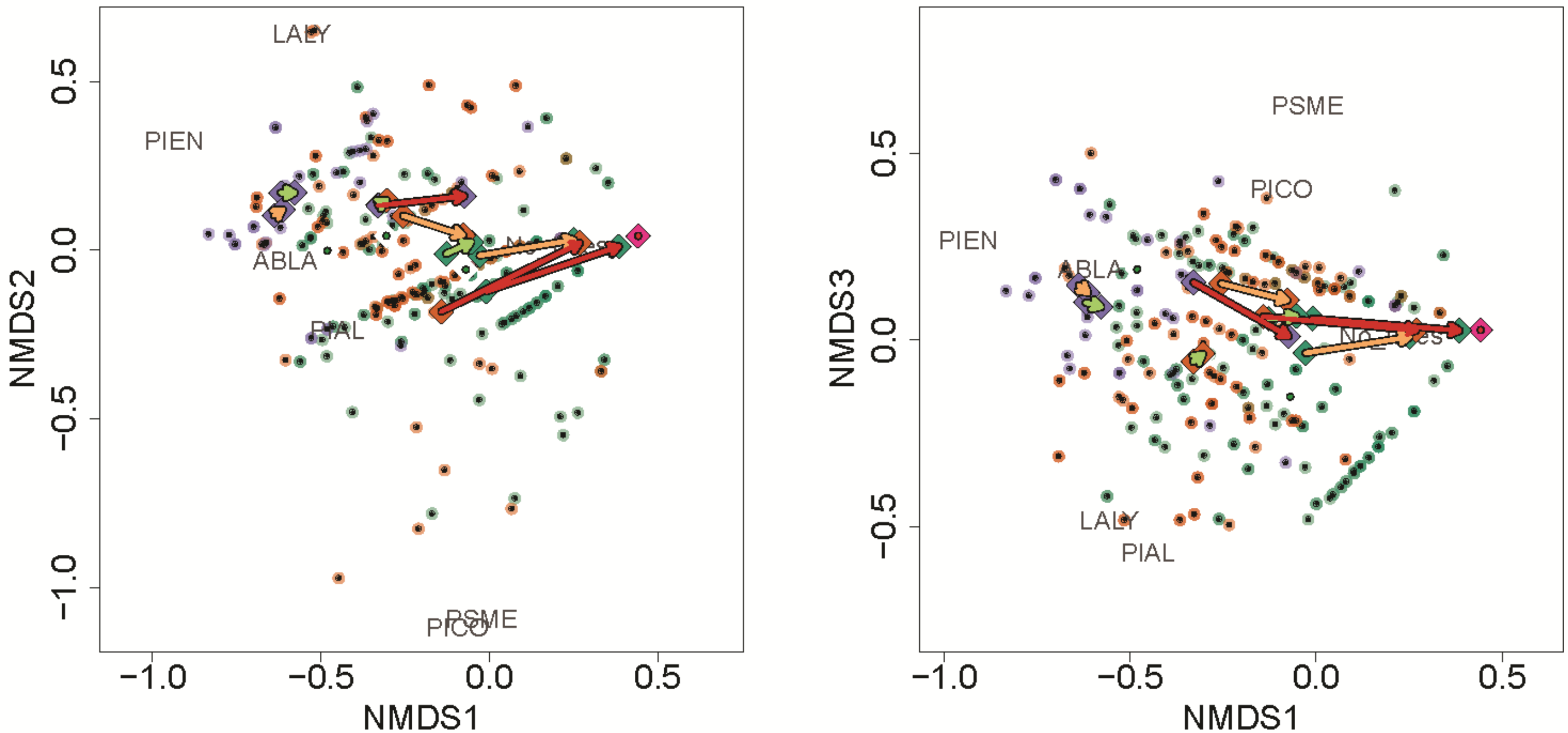


Figure 2.11b. A three-dimensional NMDS ordination of density of tree species, showing differences in relation to fire severity. Smaller circular points show locations of plots in ordination space, and are colored by canopy cover class non-forest – pink (all plots were located in same ordination space), parkland – teal; open forest – orange, closed forest – purple. Arrows connect centroids of each canopy cover class from pre-fire to post-fire, for a given level of burn severity. Arrows are colored by the burn severity class: low severity (green), moderate severity (orange), and high severity (red). The diamond-shaped starting and ending points of the arrows are colored by canopy cover class, using the same colors as for plots.

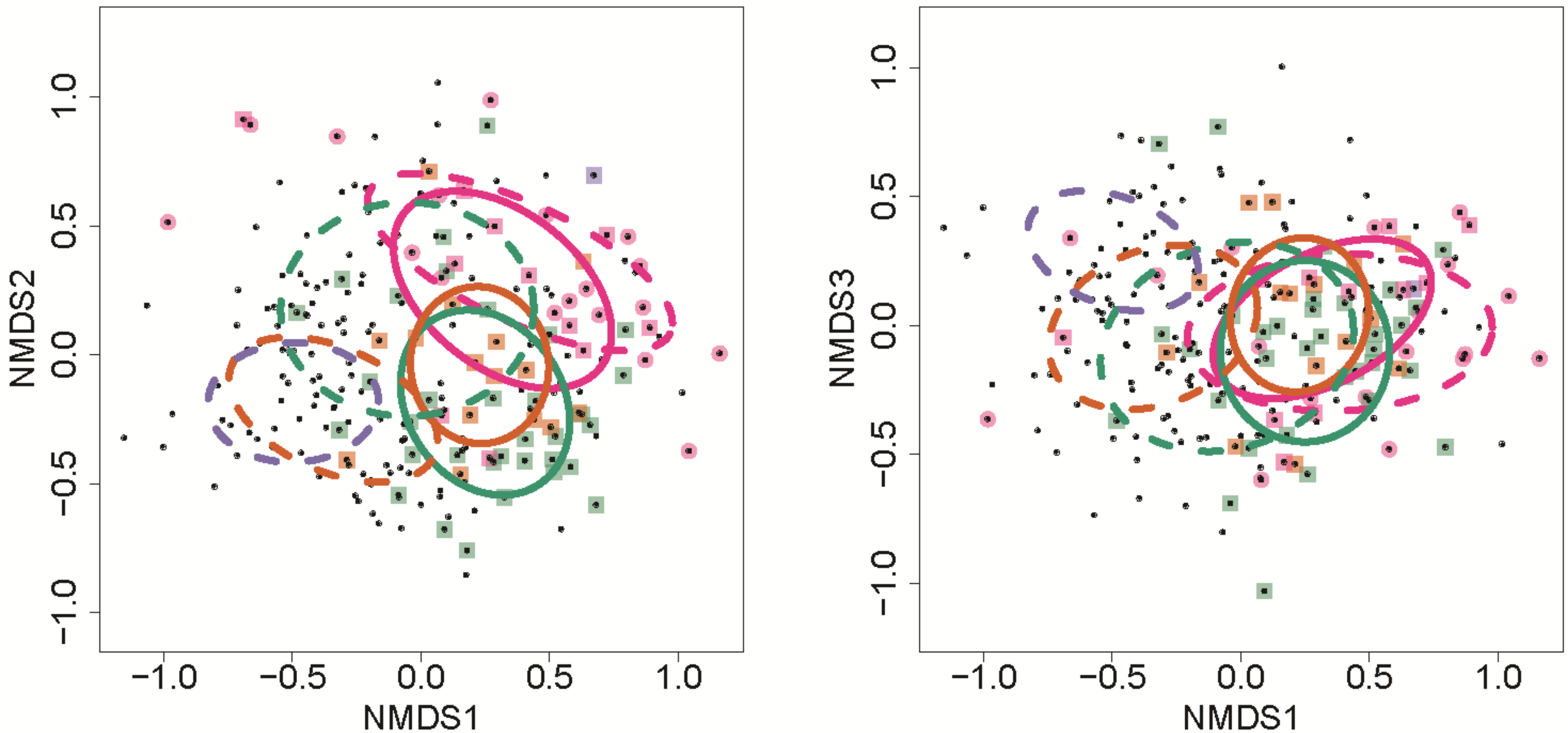


Figure 2.12a. A three-dimensional NMDS ordination of ground-layer vegetation showing locations of all plots (black points) and plots that were "non-forest" after the fire (colored points: non-forest – *pink*, parkland – *teal*; open forest – *orange*). No closed forest plots became non-forest after fire. Ellipses made with solid lines mark one standard deviation (hence not circles because of different values on the two axes) of the locations of post-fire non-forest plots by pre-fire status using the same colors. Ellipses made with dashed lines drawn around the standard deviation of the locations of unburned non-forest, parkland, open forest, and closed forest (purple) plots. The ordination had a final stress of 16.6.

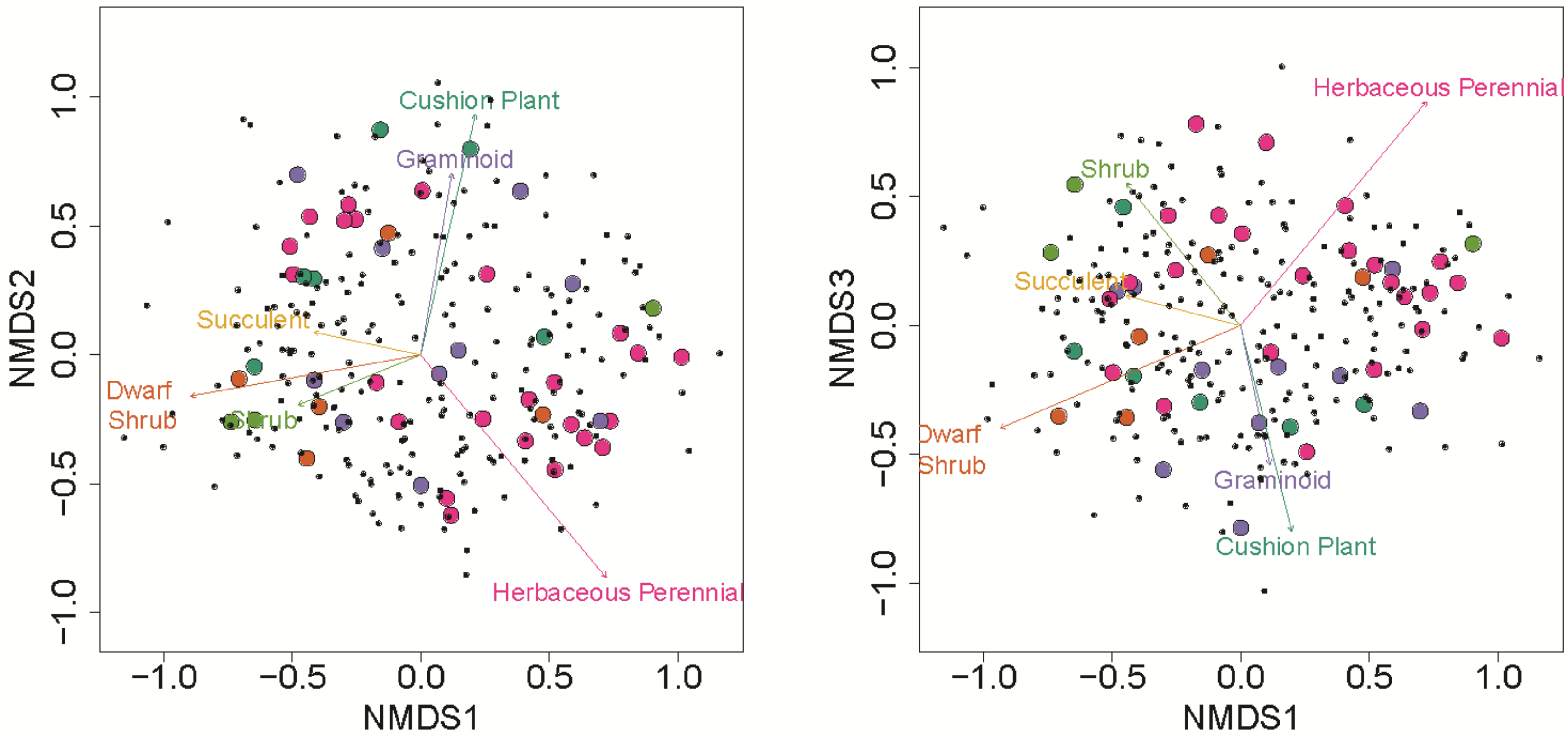


Figure 2.12b. A three-dimensional NMDS ordination of ground-layer vegetation, with small black points representing plots, and colored points showing the locations of species in ordination space. Species are colored by their growth form group; colors corresponding with the labels in the plot. Only species present in >5% of plots are shown. Arrows show the direction in which functional group cover changed most rapidly; vector length along an axis is scaled to the correlation with the axis. Arrows are plotted only where correlations were significant (P <0.05) based on a permutation test.

Appendix 2.1. Supplementary methods and results

Relationship between canopy-cover classes and other measures of overstory structure

Canopy-cover class assigned in the field was a strong proxy for other aspects of overstory structure. Cover classes captured measurable differences in total tree density and basal area ($P \leq 0.001$; Table A2.1.1, Table A2.1.2, Figure A2.1.1), and in the basal-area distribution between diameter classes (Table A2.1.1 & Figure A2.1.2). Differences were significant in both parametric and non-parametric tests. As expected, tree density and basal area increased from parkland to open forests to closed forest, although the ranges of values overlapped between classes (Figure A1). Cover classes also differed in diameter distributions: in parkland there were few trees in the largest size class (>40 cm DBH) and in closed forest there were higher densities in the two larger size classes (>20 cm and >40 cm DBH) (Figure A2). Differences between classes were smaller in the gridded plots than the structure plots, probably reflecting the strong contrast in spatial scales of measurement for structural variables (2.82 m radius) vs. cover (15 m radius). Based on models and plots, the canopy-cover classes can be seen as representing differences in tree abundance, tree basal area, and tree size structure, in addition to different canopy cover.

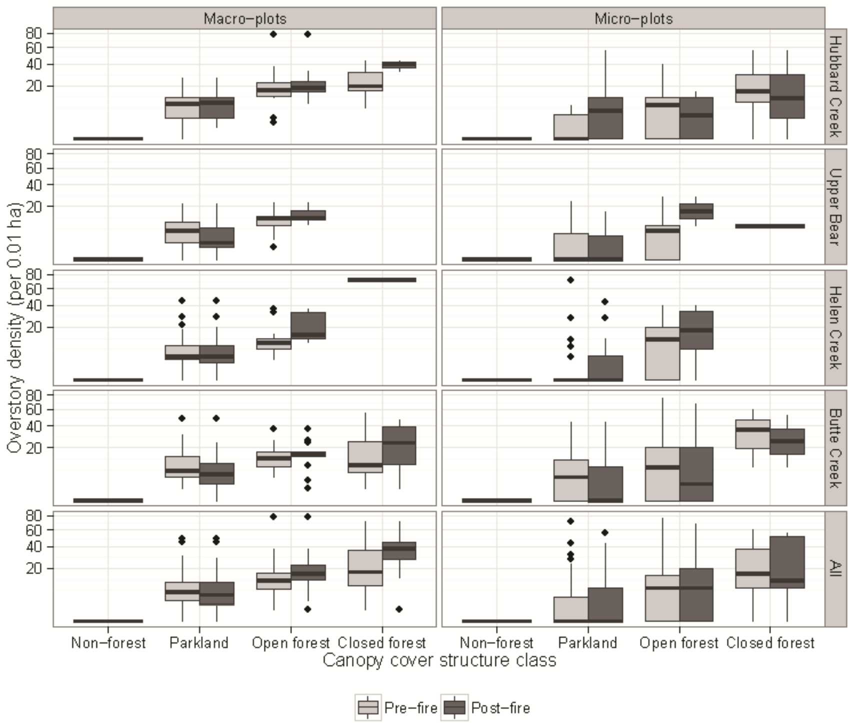
Table A2.1.1. Model results testing for differences in overstory tree density and basal area between canopy cover classes. Test statistics are for linear mixed-effect models and F for PERMANOVA.

Response variable	Model type	Transformations	Random effects or group	AIC	*d.f.*	Parkland vs. Open Forest		Open forest vs. Closed forest	
						Test statistic	*P*	Test statistic	*P*
Density	Linear Mixed Effects	Square-root	Sites within plot types	3109	770	-10.494	<0.001	-6.096	<0.001
Basal area	Linear Mixed Effects	Cubed-root	Sites within plot types	341	770	-11.516	<0.001	-7.733	<0.001
Density by size class	PERMANOVA[1]	Square-root	Fire	NA	1	24.567	<0.001	5.984	0.003

[1] Assessed only for structure plots

Table A2.1.2. Non-parametric tests for differences in overstory tree density and basal area between canopy cover class.

Response variable	**Model type**	**Data**	**Comparisons**	***W***	***χ²***	***d.f.***	***P***
Density	Kruskal-Wallis H Test	Structure plots	Parkland, Open forest, and Closed forest		73	2	<0.001
	Wilcoxon rank sum test with continuity correction		Unforested vs. Parkland	1,390			<0.001
			Parkland vs. Open forest	18,091			<0.001
			Open forest vs. Closed forest	1,652			0.001
	Kruskal-Wallis H Test	Gridded plots	Parkland, Open forest, and Closed forest		46	2	<0.001
	Wilcoxon rank sum test with continuity correction		Unforested vs. Parkland	25,715			<0.001
			Parkland vs. Open forest	17,756			<0.001
			Open forest vs. Closed forest	1,538			0.022
Basal area	Kruskal-Wallis H Test	Structure plots	Parkland, Open forest, and Closed forest		98	2	<0.001
	Wilcoxon rank sum test with continuity correction		Unforested vs. Parkland	1,390			<0.001
			Parkland vs. Open forest	20,730			0.014
			Open forest vs. Closed forest	1,480			0.000
	Kruskal-Wallis H Test	Gridded plots	Parkland, Open forest, and Closed forest		98	2	<0.001
	Wilcoxon rank sum test with continuity correction		Unforested vs. Parkland	25,715			<0.001
			Parkland vs. Open forest	17,551			<0.001
			Open forest vs. Closed forest	1,627			0.048



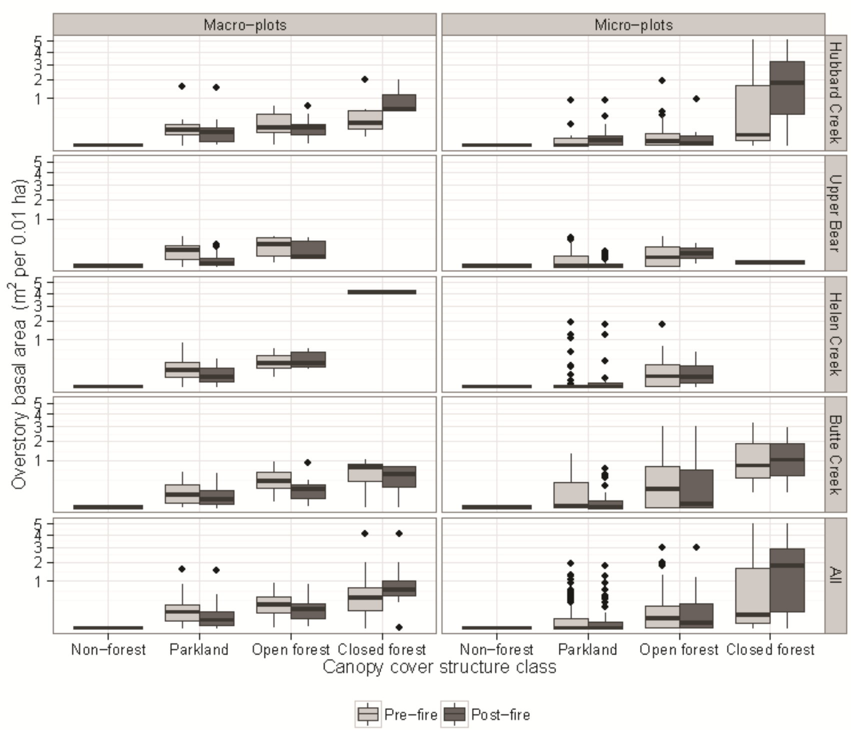


Figure A2.1.1. Box plots of tree density (top) and basal area (bottom) for the four canopy-cover classes. Each plot is included twice: once using its pre-fire canopy cover class, and the reconstructed pre-fire tree density or basal area, and once using its post-fire canopy cover class, with its post-fire tree density or basal area

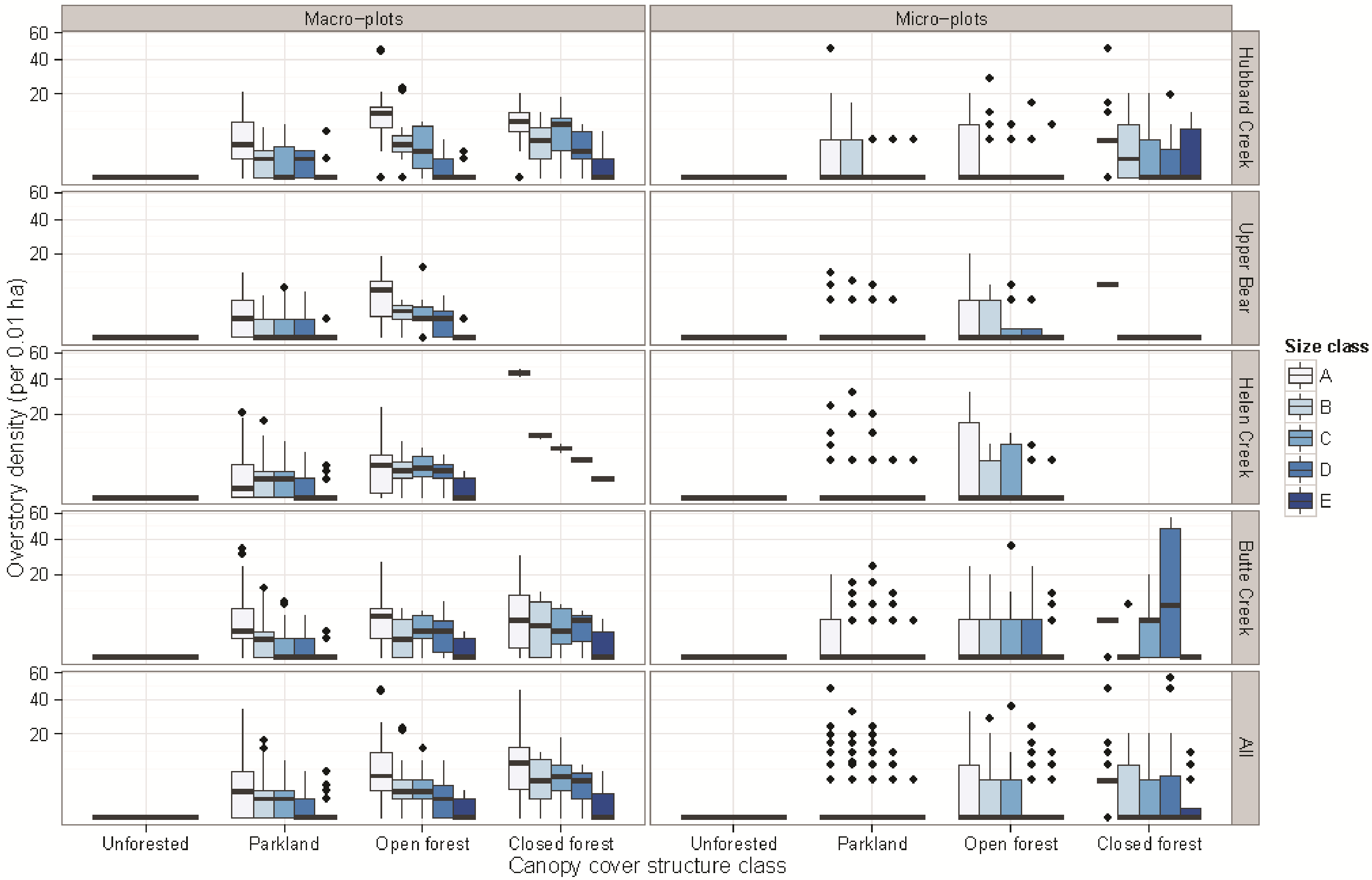


Figure A2.1.2. Box plots of tree density by size class within each canopy-cover class. Size classes are: A: <5 cm, B: ≥5 to <10 cm, C: ≥10 cm to <20 cm, D: ≥20 cm to <40 cm, E: ≥40 cm. Each plot is included twice: once using its pre-fire canopy cover class, and the reconstructed pre-fire tree density or basal area, and once using its post-fire canopy cover class, with its post-fire tree density or basal area

Table A2.1.3. Pearson product-moment correlation coefficients between original pre-fire live tree densities and pre-fire live tree densities after unidentified trees that were dead after fire but live before fire were assigned a species. To reconstruct pre-fire species composition, trees coded as unknown in the field were assigned to one of three species based on the relative abundance of live individuals within each site. Only three species could not be identified when dead: *A. lasiocarpa, Pinus contorta,* and *Picea engelmannii.* All trees in size class E were identified.

Overstory density

Size class	***A. lasiocarpa***	***P. contorta***	***P. engelmannii***
All	0.988	0.969	0.998
A	0.996	1	1
B	0.973	0.921	0.996
C	0.955	0.934	0.943
D	0.973	0.941	1
E	NA	NA	NA

Overstory basal area

Size class	***A. lasiocarpa***	***P. contorta***	***P. engelmannii***
All	0.996	0.958	1
A	0.996	1	0.999
B	0.974	0.944	0.995
C	0.947	0.934	0.965
D	0.977	0.941	0.997
E	NA	NA	NA

Appendix 2.2. Field protocol and data sheets

-
- Field Protocol, Summer 2012

C. Alina Cansler **PhD project on fire in the alpine treeline ecotone** **acansler@uw.edu**

- Last revised: 3/4/2013. Protocal jointly written by C.A. Cansler and S. Hiebert (crew lead).
-
- **What to bring in the field**
- 30-meter tape, height pole, 5-meter cloth diameter tape, 1 "executive" diameter tape per person, 1 compass per person, 1 small rite-in-rain notebook per person, camera, laser range finder, GPS unit, radio, extra batteries, pencils, clipboard, tatum, data sheets, first aid kit, white notebook, plant book(s), clinometer, maps of plots, topo map.
- Warm clothing, raingear, sun hat, sunscreen, bug spray, and food for the day.
-
- **Field units**
- All measurements will be taken or converted to metric units. The International System of Units is a modernized version of the metric system established by international agreement that provides a logical and interconnected framework for all measurements in science, industry, and commerce. As such, metric is the only acceptable standard for all scientific endeavors and will be the only acceptable units for this research program.
-
- **Summary of "macro-plot" (belt transect plot) data collection**
- **General plot set up**
- Locate plot
    - Select plot from GPS and navigate to plot (technically the starting point of the plot transect). You should be less than 2 meters from the point, and preferably at 0 or 0.5 meters from it, as show on the GPS unit.
- Set up plot
    - Put a pin in at one end of the tape at the point the GPS leads you to. This is the plot start point.
    - Use your compass to determine aspect at the plot start point
    - Record plot number and aspect on data sheet (on the seed source diagram).
    - The plot azimuth is 90 degrees *counterclockwise* (to the *left*) of the plot aspect. Adjust your compass, shoot the azimuth with your compass, and then run the 20 meter plot transect along this azimuth.
    - Secure the far end of the tape so the tape is tight and straight.
- GPS plot
    - Record the GPS coordinates of the center (at 10 meter) of the plot transect.
    - Label the point with (1) the prefix for the fire (bc = Butte Creek, hc = Hubbard Creek, lb= Little Bear, hl = Helen Creek) and the plot number you navigated to.
    - Average the plot location for long enough that the accuracy is <3.5 meters. (The GPS unit can be left while other data is recorded).
    - Stop averaging and save the GPS point.

  - Record GPS coordinates, elevation, *and* the error on data sheet.
- Plot pictures
  - Make sure time on camera matches the GPS unit. Remember to switch it if we change time zones.
  - Plot pictures should be taken in four directions from plot center, starting with upslope.
  - Record plot pictures and camera name on seed source assessment diagram.
  - Both horizontal and vertical pictures are acceptable; the goal is to capture the plot conditions.
  - Occasional "action shots" of folks collecting data are also good to take ;-)

- <u>Coversheet</u>
- **Fire:** The name of the fire.
- **Plot number:** A unique number for each plot, from the GPS unit.
- **Date:** MM/DD/YYYY that plot was sampled.
- **Time:** 24-hour time the plot was sampled.
- **Team:** Names, or name codes (first two letters of first and last name) of team that sampled the plot. Circle the coversheet record's name.
- **UTM:** UTM zone (example "10U").
- **Easting:** UTM easting coordinates (as saved, after averaging in the GPS unit).
- **Northing:** UTM northing coordinates (as saved, after averaging in the GPS unit)
- $\pm$ GPS error, as shown after averaging in the GPS unit/
- **Elevation:** GPS elevation, in meters (as saved, after averaging in the GPS unit)/
- **Tree flagging? winds from?** Compass direction of the direction the winds usually come from; only for plots that there is flagging. On the plots without flagged trees, record "N/A."

Figure removed in order to avoid copyright infringement.

Figure: Example of tree flagging

- **Soils:** Options are: Hydric (in wet depressions), Humus, Eroded (for places that likely had humus soil before fire, but are now eroded), Lithosols, Beadrock

Figure removed in order to avoid copyright infringement.

Figure: Where different alpine soils occur


Figure: Hydric soil examples


Figure: Alpine humus examples


Figure: Eroded soil examples


Figure: Lithosol examples

Figure: Bedrock examples

- **Tree clump:** If present record tree clump type. Options: Individual, Circular, Flag, Triangular, Streamline, Elongate, Krummholz. If not present record NA

  Figure removed in order to avoid copyright infringement.
  Figure: Tree clump types

- **Notes**: impacted by soil creep, avalanches, snow movement, gophers, ungulates, older fires, bark beetles, etc.) It is OK to also include any information that made the plot memorable: For example, "Clark's nutcracker sighting on plot at 2:34 p.m."

- **Severity**
- Estimate either the percent cover or percent consumed for each 5-meter plot section of each of the following:
    - Pre-fire cover litt/moss/duff
    - LITT/MOSS/DUFF consumed
    - Pre-fire cover shrubs/herbs
    - Shrubs/herbs consumed
    - Pre-fire tree canopy cover
    - Post-fire tree canopy cover
    - Pre-fire logs cover
    - Pre-fire logs charred
- One easy way to think about percent cover is that every square meter equals percent of the plot. Another way to visualize percent cover is by the figure on the right.
- Remember the natural tendency is to over-estimate percent cover.
- **LOC**: Your level of certainty regarding the severity estimate. L= low, M=moderate, H=high.

Table: areas as percent of plot

Total plot (50 m2)			1/4 section (25 m^2)			
m2	**%**	**Side of sq.**	**m2**	**%**	**Side of sq.**	Figure removed in order to avoid copyright infringement.
0.5	**0.5**	**0.71**	**0.125**	**0.5**	**0.35**	
1	**1**	**1.00**	**0.25**	**1**	**0.50**	

5	**5**	**2.24**	**1.25**	**5**	**1.12**	
10	**10**	**3.16**	**2.5**	**10**	**1.58**	
25	**25**	**5.00**	**6.25**	**25**	**2.50**	
50	**50**	**7.07**	**12.5**	**50**	**3.54**	

•

Component and overall burn severity

- Component and overall burn severity are assessed at each of the following scales, from the center (10m on transect) of the plot: 2.82 m radius circle, 20 m by 5 m (macro-plot area), 15 m radius circle, 60 m radius circle.

Table. ATE "Components"

Sub-nival	Vegetation is patchy and restricted to only the most favorable locations. Rock & talus substrates
Alpine (meadow)	Closed carpet of vegetation that includes alpine meadows, shrubs and sporadic dwarfed trees
Krummholz	Short stature, bent twisted trees on all or some of the plot
Parkland	5-10% tree canopy cover; if trees occur in clumps, note clump form
Open forest	10-40% tree canopy cover
Closed forest	> 40% tree canopy cover

- Possible burn severities are unburned (U) (no evidence of fire anywhere, very low (VL) (low burn severity; only small portion of area effected), low (L) (surface fire, few, if any trees killed), moderate (surface fire with occasion consumption of individual trees, between 20-70 % trees killed), high (continuous surface fire with torching, or carried through the crown; 50-100% of trees killed), very high (likely crown fire, all trees killed, and surface fuels consumed; little organic soil persisted through the fire).

Seed Source Assessment

- To determine the source of seedling regeneration. From the center of the plot (x=10, y=0), orient yourself down slope and determine the aspect. This will be the aspect of the dashed line on the coversheet. Determine the angles of intersections of the quadrants by adding 45 to your first aspect. Then add 90 to each subsequent aspect. For example, if down slope is 20 degrees on a specific plot, then the first quadrant would begin at 65 degrees and the other aspects would be 155, 245, and 335 respectively.
- **Observer** – name or name-code for observer. *Very important* – we will need your eye-height calculate the tree-height.
- Set laser on "horz" setting for distance calculation, and "inc" setting for slope and height measurements.
- Check and make sure units "inc" is *degrees* and units for "horz" is in *meters*.
- The closest three tree seed sources in each quadrant will be recorded.
    - **Spp**. – 4-letter tree species code
    - Horizontal distance in meters, as measured by the laser range finder. If tree is very close (<3 meters, measure distance by holding tape horizontal.) The laser range finder should be set on the horizontal distance setting. On this setting, you can hold the laser at any angle and it will give you the distance along a 0 degree slope to the object. This way, you can take a shot anywhere you have a clear view of the bole of the tree, unobstructed by tree branches.
    - **B/A/S** – did the tree *establish* before the fire (most cases), after the fire, or is it a dead tree with serotenous cones (dead PICO)? If A or S, then record extra B trees for that quadrant in blank rows.
    - **Slope** – record slope, to your eye-height above the base of the tree (in degrees for all fires but Butte Creek, where we used percent)
    - **Treetop** – record angel to the top-most branch of the tree
    - **Height** – only needed if tree was too close to center of the plot for the tree top angel to be measured accurately. (appx. >150 deg). In that case, measure the height with the laser range finder, from a location that you can be further away from the tree, and get a clear view of the top (and your eye height on the trunk) and record the height from the rangefinder. Measure height to your eye height on the bole of the tree, not to the base of the tree.

Facilitator transect

- The purpose of this transect is to get the distribution within the plot of things that would count as facilitators for seedling trees. Since we count facilitators when they are within 0.03 meters of a seedling (see seedling assessment, below) we also want the null distribution of the area within 0.03 meters of a seedling.
- **Fire:** The name of the fire
- **Plot number:** A unique number for each plot, from the GPS unit
- **Recorder:** crew who is collecting the data, circle the recorder's name
- **T.D. start**: The distance along the transect where the facilitator, or the area "near" (within 0.03 meters) the facilitator starts, to the nearest 0.05 meters
- **T.D. end**: The distance along the transect where the facilitator, or the area "near" (within 0.03

meters) the facilitator ends, to the nearest 0.05 meters. *Should always be larger than T.D. start distance!*

- **Type/Spp.** What type of facilitator is it?

Table: Possible facilitators and their codes

Code of facilitator	Code of area within 0.3m of the facilitator	Description
LO	NLO	Log (>10 cm in diameter where the transect crosses the log)
BO	NBO	Boulder (25.6 409.6 cm)
BED	NBED	Bedrock (>409.6 cm)
STMP	NSTMP	Stump (< 1.4 meters tall, and > 10 cm diameter)
SNAG	NSNAG	Snag (>= 1.4 meters tall, and > 10 cm diameter)
ABLA	NABLA	Tree bole (>10 cm diameter at base) (ABLA in this example; use other tree codes as appropriate)

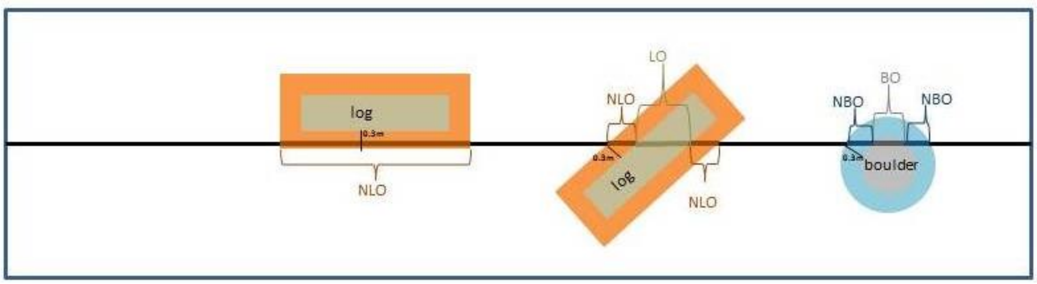


Figure: What is "near" a log or boulder

- Remember, you can't just add 0.03 meters on to each side of facilitators that are crossed by the transect line. In the above example, the line is within 0.03 meters of the first log, even though it doesn't cross it. Also, if a log is not at right angles to the line, then the "NLO" area will be > 0.03 meters.
- How to deal with overlapping facilitators:
    - If they are completely different types (a log and a boulder, for example) then their distances along the transect can overlap.
    - if the facilitators are of the same type then their distances should not overlap:
        - Example 1: Boulder from 0.50 meters to 0.7 meters, and another boulder from 0.65 to 0.80; enter BO 0.50 to 0.70 and BOLD 0.70 to 0.80 *or* enter BO 0.50 to 0.80.
        - For logs, it is necessary to have two separate entries, so the diameter of each log can be entered:
            - Example 2: log from 0.50 meters to 0.7 meters, and another log from

0.65 to 0.80; enter LO 0.50 to 0.70 and LO 0.70 to 0.80. It doesn't matter where, exactly, the distance along the tape that one log stops and another starts, but measure the diameters correctly.

- Facilitators (logs, boulders, bedrock, etc.) take precedence over "near" that *same* facilitator.
  - Example: If a "log" overlaps a "near log" area of different log, only record the "log".
  - Example 2: If a "near boulder" overlaps a "boulder" area of different log, only record the "boulder".
  - Example 3: "Near boulder" *can* overlap a "near log" or overlap a "log", since a boulder is a different type of facilitator than a log.

- **DBH:** For each log, record its diameter at the point where the transect line crosses the central line of the log. (Note that this is actually not the DBH, the diameter, despite the label of this column on the data sheet). Do *not* record the distance of the transect line over the log – that is already being captured by the **T.D. start** and **T.D. end**
  - Log diameters: if possible measure with a diameter tape, but if log is on the ground and you can't get the tape under, look vertically down on side of the log and set the tape to 0, and then hold the tape in the middle of the log, and look vertically down the other side to get an accurate diameter measurement (to the 0.5 cm).
- **Decay:** For each log, record the decay class, following the descriptions below in the "**Log decay classes" figure**
- **B/A:** For each log, did it fall to the ground before or after the fire? Usually, if the plot was burned, it is fairly obvious if the log was present before the fire: the charring will be horizontal to the ground, often along the bottom surface of the log. Logs that were standing when the fire burned will either be charred near the break from the stump or at the roots.
- **L/Snag Class:** For live trees, L = Living; for dead trees, RC= Recent; green or brown needles present; FR = Fire; no needles present, evidence of fire but sapwood/heartwood NOT deeply charred; BC = dead before fire; sapwood/heartwood deeply charred; BN = dead before fire; no char; note evidence in comments
- **LOC:** your level of certainty for B/A or Snag Class (L=low, M=medium, H=high)

Figure removed in order to avoid copyright infringement.

Log characteristics	**Log decomposition class**				
	1	**2**	**3**	**4**	**5**
Bark	intact	intact	trace	absent	absent
Twigs <3cm (1.18 in)	present	absent	absent	absent	absent
Texture	intact	intact to partly soft	hard, large pieces	small, soft blocky pieces	soft and powdery
Shape	round	round	round	round to oval	oval
Color of Wood	original color	original color	original color to faded	light brown to faded brown or yellowish	faded to light yellow or gray
Portion of log on	log elevated on	log elevated on	log is sagging	all of log on	all of log on

ground	support points	support points but sagging slightly	near ground	ground	ground

Figure: Log decay classes fig. 44 and table 20, p. 80 of Maser et al. 1979).


Figure: Example of logs that were likely on the ground before the fire. On left, note that char is only along the base of the log: strong evidence that it was down before the fire. On the right, the whole logs are charred, so they were either down, or standing dead trees at time of fire. Inspection of where the logs broke from the base of the tree is needed here: if that area is un-charred, with white splintered wood at the break, then it fell after the fire (Log class of "A"); if the break is deeply charred, it likely fell before the fire (log with "B" class).

<u>**Vegetation Transect Assessment**</u>

- Along the transect line we will use the point-intercept method to assess the vegetation in the plot. "The theory behind this method is that if an infinite number of points are placed in a two-dimensional area, the exact cover of a plant species can be determined by counting the number of points that hit that species. This method then estimates the values from the infinite number of points through the use of a sample number of points" (from FMH handbook)

  Figure removed in order to avoid copyright infringement.

  Figure 8.10 from Chapter 8 of Elizinga et al. 1998: point intercept method (from Measuring and Monitoring Plant populations)

- At each point we will use a sampling rod (a lightweight pole), placed vertically determine the specific point. *It is very important to place the pole plum to the ground - Sampling errors can occur if it is not lowered plumb to the ground!*

- We are measuring two things at each point (1) the substrate covering the ground and (2) the type and height of the vegetation covering the ground.
- **Substrate:** record the inorganic surface under the base of the pole; if the base of the pole is on the rout-crown of a plant, record that instead of the leaf. Do not record plants as substrate

if you are only pinning a leaf, and not the place where the plant roots to the ground.

Figure removed in order to avoid copyright infringement.	**Figure 8.7 from Chapter 8 of Elizinga et al. 1998: Example of the "root crown" of the plant; if the base of the pole land on this part of the plant, record the plant as substrate, and also record it again in the cover category.**

- **Height:** record the highest height class where any plant touches the pole. Height classes are as follows: <0.01, <0.05, 0.05-0.1, and then up in 0.1 meter segments.
- **Species:** record the species codes for the tallest species touching the pole first, and then record all other species that touch the pole. Also record any species that are directly under the pole, including species recorded as "substrates"
- We will take data at each point down the transect line beginning at 0.5m at intervals of 0.5m until the 20 meter point at the end of the transect.
- **Unknown species**: if you do not know that species record the first two letters of the genus and either XX or YY depending on which field crew you are in, and then a number. Take picture; or the whole plant and of important floristic characteristics (flowers, back of leaves, ligules on grasses, etc.). *Be sure to record the code, a description, picture numbers and plot numbers in the "white notebooks"!!!*
- Trees and seedlings are not recorded in the point intercept transects, only shrubs, herbs, and substrates

<u>Seedling Assessment</u>

- Seedlings are sampled within 2.5 meters on each side of the 20 meter transect. The 2.5 meters is in horizontal distance (not along the slope) from the transect line.
- Use the pole to determine which trees are in and out; if close measure with the tape
  - The center of the seedling (i.e. the pith of the tree) must be in for the it to be counted. This means that a stem rooted inside the plot but leaning outside should be counted as "in."
  - If the tree is directly on the line, it will only be counted if it is on the downslope edge of the transect; directly on the line but on the upslope side will be counted as out. In almost all cases the center pith of the tree will either be in or out, so this should rarely come up.
- On the rare occasions where there are an extremely high abundance of seedlings in a plot (over 50 in the first 5 meters, and this seems consistent throughout the plot area, the plot may be subsampled. (1) Confirm with Alina on the radio about this. (2) sample a belt along the one side of the transect, 1.25 uphill from the transect by 20 meters long.
- I recommend making a U shape by starting at one end and going length-wise down one half of the plot and then making a U turn and going up the other half.
- For this study, a seeding is considered a tree that is < 1.4 m in height.
- In Krummholz growth form situations, where adult trees are < 1.4 m tall, most of the trees appear to consist of stem clumps. Ideally each stem should count as a tree and be assigned clump numbers and letters as appropriate. In dense Krummholz trees, however, stems may be

very difficult to trace to the tree base. In the interests of time under these circumstances, each dense clump may be counted as a single tree and examined as thoroughly as possible.

- Clumps: Stems whose bases occur outside the plot are not included; it is thus possible to have some stems of a clump counted and others not. If a tree forks and the stems are separate below breast height, each stem is measured and evaluated separately.

Figure removed in order to avoid copyright infringement.

- **Figure: Identifying tree clumps (mostly for PIAL) (Figure 3-4 from Greater Yellowstone Whitebark Pine Monitoring Working Group 2011).**
-

- **Fire:** The name of the fire
- **Plot number:** A unique number for each plot, from the GPS unit
- **Recorder:** crew who is collecting the data, circle the recorder's name
- **X:** the section of the transect the seedling is in: 0-2.5 meters, 2.5-5, 5-7.5, 7.5-10, 10,12.5, 12.5-15, 15-17.5, 17.5-20
- **Y:** enter "+" if seedling is on "upslope" (left) side of the transect line, and enter "—" if seedling is on downslope (right) side of the transect line.
- **Species:** four letter tree species code
- **Height**: Height class of the highest live branch of the seedling.
  - Height classes are as follows: <0.01 (recorded as .1), <0.05 (recorded as 0.05), 0.05-1.5 (recorded as 1), and then up in 0.1 meter segments, with the last segment at 1.35 to 1.40 (recorded as 1.4).
  - Note that the seedling height pole is already "averaging" for you. For example the segment from 0.75 to 0.85 meters is labeled "0.8".
  - If seedling is prostrate, measure along the length of main stem of the seedling. If there is no obvious main stem (a really brushy Krummholz growth form) then measure the mean height of the uppermost branches.
- **1st year:** whether or not it is in its first year of growth (designated by Y=Yes, N=No
- **B/A**; whether the seedling regenerated before or after the fire (designated by B=Before fire, A=After fire)
- **LOC**: Level of Certainty regarding before or after the fire (L = low, M = moderate, H = high)
- **Facilitators**: type and distance to the closest three facilitators; record the closest one first, next closest second, etc. If more than 3, just record the closest 3.
- Don't record dead seedlings

Table: facilitator type

Code of facilitator	Description
LO	Log (>10 cm in diameter where the transect crosses the log)
BO	Boulder (25.6 409.6 cm)
BED	Bedrock (>409.6 cm)
STMP	Stump (< 1.4 meters tall, and > 10 cm diameter)

SNAG	Snag (>= 1.4 meters tall, and > 10 cm diameter)
ABLA	Tree bole (>10 cm diameter at base) (ABLA in this example; use other tree codes as appropriate)

Figure: seedling within 0.03 meters of log facilitator (left), and of snag (right)

- **Damage codes**; record any damage on seedlings or trees; seed description after overstory tree assessment below.
-

Overstory Assessment

- The information we are collecting for the overstory assessment allows us to do two things:
    - Record current conditions
    - And *reconstruct* pre-fire conditions (live and dead trees that were standing before the fire)
- Overstory is sampled within 2.5 meters on each side of the 20 meter transect. The 2.5 meters is in horizontal distance (not along the slope) from the transect line.
- Use the pole to determine which trees are in and out.
    - If uncertain measure distance with the cloth tape.
    - The center of the tree bole (i.e. the pith of the tree) must be in for the tree to be counted. This means that a stem rooted inside the plot but leaning outside should be counted as "in."
    - If the tree is directly on the line, it will only be counted if it is on the downslope edge of the transect; directly on the line but on the upslope side will be counted as out. In almost all cases the center pith of the tree will either be in or out, so this issue should rarely come up.
    - For dead and down trees, the center of where the tree was rooted *before the fire* must be in.
    - Uprooted dead trees with no obvious rooting location (usually smaller trees, DBH <5cm) should not be counted: they may have been moved into the plot by snow movement, etc.
- *Never subsample the overstory. Always measure the whole plot!*
- If a "trunk" splits off the main bole below DBH but clearly above the root/base, it should be considered a branch of the tree and not a separate stem. These branches often curve outwards and upwards. The same rule applies to groups of stems: Stems that are separate below breast height, although they occur within a group or "tree clump," are each evaluated:

    Figure removed in order to avoid copyright infringement.

-
- **Figure: Distinguishing a branch from a tree (Figure 3-4** from Greater Yellowstone Whitebark Pine Monitoring Working Group 2011).
-
- **X:** the section of the transect the overstory tree is in: 0-2.5 meters, 2.5-5, 5-7.5, 7.5-10, 10,12.5, 12.5-15, 15-17.5, 17.5-20
- **Y:** enter "+" if tree is on "upslope" (left) side of the transect line, and enter "—" if tree is on downslope (right) side of the transect line.
- **Status =** L = Living; D = Dead; F = Fallen; if the tree was *standing* (either live or dead) before the fire, it should be recorded, even if it is on the ground now.
- **Snag-Log class:** (S-L class) RC= Recent; green or brown needles present; FR = Fire; no needles present, evidence of fire but sapwood/heartwood NOT deeply charred; BC = dead before fire; sapwood/heartwood deeply charred; BN = dead before fire; no char; note evidence in comments (Note: on the Tyee Fire, classes were as follows: A = green or brown needles present; B = no needles present, sapwood/heartwood NOT charred; C = no needles present, sapwood/heartwood charred)

Figure: Left: tree at center with deeply-cared base was likely dead at time of fire. Status = D, S-L class = BC. Tree at right likely alive before fire. Status = D, S-L class = FR. Right: close-up of deeply charred tree that was likely dead before the fire.

- **LOC**: Level of Certainty Snag class(NA = not applicable (for live trees) L = low, M = moderate, H = high)
- **DBH:** diameter at 3.7 meters of the tree
- **L/Snag Class:** For live trees, L = Living; for dead trees,
- **B/A:** did *live* tree establish before or after the fire?
- **LOC:** your level of certainty for B/A
- **Damage codes:** see description in seedling assessment, above.
- *Pay particular attention to white pine blister rust on the PIAL: <u>check every PIAL</u> for it, don't just note it if you see it!!*
- For recently dead trees used damage codes to describe the factors of death. Blister rust cankers and the galleries of mountain pine beetles are often still evident in recently dead trees. Notes may be made on data sheets in order to elaborate on any tree condition.
    - **WPBR:** Determine if branches and stems have cankers. If so, determine if the cankers are active or inactive.
    - **Presence/absence of mountain pine beetle**: Mountain pine beetle infestation is another major mortality factor in whitebark pine. Either beetle entry holes with pitch plugs or J-shaped galleries in the wood indicate their presence in a tree. Removal of a section of stem bark will be necessary to view beetle galleries.

Table: General codes

CLUMP	same location (seed cache) (note trees it is grafted to)	GRAFT	boles grafted together <1.4m (note trees it is grafted to)
STOP	Stunted Top	KRUM	Krummholz (record area in m^2)
DTOP	Dead Top	LAYR	Layering tree branches (record area in m^2)


Figure: Example of a clump of 10 PIAL seedlings (left) and layering ABLA (right)

Table: Physical Damage Codes

SD	Snow Damage (crushing)
CT	Crushed by fallen Tree
AD	Avalanche Damage
BTOP	Broken Top
SM	Snow Mold

Table: bark beetle codes - Record any other beetle species in notes

Bark Beetle Codes	
BBG	Bark Beetle Galleries
BBS	Bark Beetles Seen
BBP	Bark Beetle Frass/Pitch/Holes
MPB	Mountain pine beetle *(D. ponderosae)*
SB	Spruce Beetle *(D. rufipennis)*



-
- **Figure: Recient PIAL snag, not killed by fire (left). Bark beetle galeries on same snag (center). Reciently attact PIAL with entry/exit holes and pitching (right)**

Table: WPBR Codes (only PIAL)

SCA	Stem Canker Active

SCI	Stem Canker Inactive
BCA	Branch Canker Active
BCI	Branch Canker Inactive
BF	Branch Flagging

Also note any rodent chewing, or other evidence of WPBR, in the notes

Figures: Evidence of white pine blister rust (WPBR): active stem cankers (orange or yellow sporulation aecia) (SCA).

Figures: Evidence of white pine blister rust (WPBR). Left: stem canker inactive (no sporulation aecia) (SCI). Right: branch canker inactive (BCI)

Figures: Top left and right: branch flagging, which, upon closer inspection was due to WPBR. Bottom left: *Not* branch flagging. These Krummholz PIALs probably have "sun scorch" from having branches exposed above show in bright cold conditions. Bottom right: *Not* branch flagging. Whole trees are turning red from mountain pine beetle attack.

- **<u>Micro-plot assessment</u>**
- Micro-plots are a circular 2.82 meter radius plot. This is equivalent to 0.025 ha, or ¼ of the larger plot area.
- Locate plot
    - Select plot from GPS and navigate to plot (technically the starting point of the plot transect). You should be less than 2 meters from the point, and preferably at 0 or 0.5 meters from it, as show on the GPS unit.
- Set up plot
    - Put a pin in at plot center location, and extend the tape out 2.82 meters (horizontal, not slope distance).
    - It helps to have one person hold the center one end of the tape, and have another walk around to determine if trees or features (boulders, burned areas, etc.) are in or out of the plot.
    - Record plot number on data sheet.
- GPS plot: same as large plots
    - Plot pictures: same as large plots
- **Fire:** The name of the fire.
- **Plot number:** A unique number for each plot, from the GPS unit.
- **Date:** MM/DD/YYYY that plot was sampled.

- **Time:** 24-hour time the plot was sampled.
- **Team:** Names, or name codes (first two letters of first and last name) of team that sampled the plot. Circle the coversheet record's name.
- **UTM:** UTM zone (example "10U").
- **Easting:** UTM easting coordinates (as saved, after averaging in the GPS unit).
- **Northing:** UTM northing coordinates (as saved, after averaging in the GPS unit).
- $\pm$ GPS error, as shown after averaging in the GPS unit.
- **Elevation:** GPS elevation, in meters (as saved, after averaging in the GPS unit).
- **Tree flagging? winds from?** compass direction of the direction the winds usually come from; only for plots that there is flagging. On the plots that there is no flagging, record "N/A."
- **Soils:** Options are: Hydric (in wet depressions), Humus, Eroded (for places that likely had humus soil before fire, but are now eroded), Lithosols, Bedrock.
- **Herb/shrub cover:** percent cover of herbs and shrubs.
- **Height:** average height of herbs and shrubs.
- **Major species:** one or two herb or shrub species that have the greatest cover.
- **Facilitator cover:** percent cover of **trees, logs, boulders and bedrock.** This is only the cover of these objects, *not* the area that is near (within 0.03 meters of) them.
- **Notes**: impacted by soil creep, avalanches, snow movement, gofers, ungulates, older fires, bark beetles, etc.)

Component and overall burn severity

- Component and overall burn severity are assessed at each of the following scales, from the center (10m on transect) of the plot: 2.82 meter radius circle, 15 meter radius circle, 60 meter radius circle. Possible components are:

Subnival	Vegetation is patchy and restricted to only the most favorable locations. Rock & talus substrates
Alpine (meadow)	Closed carpet of vegetation that includes alpine meadows, shrubs and sporadic dwarfed trees
Krummholz	Short stature, bent twisted trees on all or some of the plot
Parkland	5-10% tree canopy cover; if trees occur in clumps, note clump form
Open forest	10-40% tree canopy cover
Closed forest	> 40% tree canopy cover

Possible burn severities are unburned (U) (no evidence of fire anywhere, very low (VL) (low burn severity; only small portion of area effected, low (L) (surface fire, few, if any trees killed), moderate (surface fire with occasion consumption of individual trees, between 20-70 % trees killed), high (continuous surface fire with torching, or carried through the crown; 50-100% of trees killed), very high (likely crown fire, all trees killed, and surface fuels greatly consumed; little organic soil persisted through the fire).

Severity

- Estimate either the percent cover or percent consumed for each 5-meter plot section of all of each of the following:
    - Pre-fire cover litt/moss/duff

  - LITT/MOSS/DUFF consumed
  - Pre-fire cover shrubs/herbs
  - Shrubs/herbs consumed
  - Pre-fire tree canopy cover
  - Post-fire tree canopy cover
  - Pre-fire logs cover
  - Pre-fire logs charred
- One easy way to think about percent cover is that every square meter equals percent of the plot. Look at percent cover handout for more information how to calculate percent cover. Another way to simplify percent cover is by the following figure. Remember the natural tendency is to over-estimate percent cover, so use the following as a guide

Seedling and tree assessment

- For each species in each height (seedlings) or DBH (overstory) classes tally the number of trees
  - Seedling height classes (m): 0-0.05, 0.05-0.1, 0.1-0.4, 0.4-0.9, 0.9-1.4
  - Overstory DBH classes (cm): <1, 1-2.5, 2.5-5, 5-10, 10-15, 15-20, 20-30, 30-40; >40 cm, record the exact DBH
- Record your tally using the dot tally system.
  1. Starts your count (1, 2, 3, and 4) with dots to create the four corners of a square.
  2. Counts 5, 6, 7, and 8 create the sides of the square by connecting the dots from counts 1-4.
  3. Counts 9 and 10 connect the diagonal dots creating an "X" inside the square.

Record the total the number of seedlings or overstory trees before leaving the plot!

References

Elizinga, C.L., Salzer, D.W., & Willoughby, J.W. 1998. *Measuring and Monitoring Plant Populations*. BLM Technical Reference. 1730-1.

Greater Yellowstone Whitebark Pine Monitoring Working Group. 2011. Interagency Whitebark Pine Monitoring Protocol for the Greater Yellowstone Ecosystem. Version 1.1. 131.

Maser, C., Anderson, R.G., Jr, K.C., Williams, J.T., & Martin, R.E. 1979. *Dead and Down Woody Material of Oregon and Washington*.

Fire in the Alpine-Treeline-Ecotone **Plot length** = 20 m (slope dist.); **Plot width** = 5 m (horizontal dist.)

Fire ____________ **Plot #**_____________ **Date**______________ **Time:**_______ **Team:**______________________________

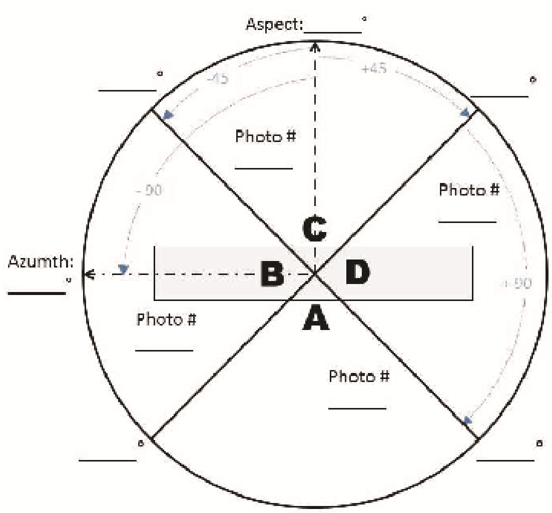
UTM:_____ **Easting**_____________ **Northing**___________ **±** ______ **Elevation**_________ **Tree flagging? winds from**___°

Soils: Hydric Humus Eroded Lithosols Bedrock **Tree clump:** N/A Individual Circular Flag Triangular Streamline Elongate Krumholtz

Severity	**1 (0-5m)**		**2 (5-10m)**		**3 (10-15m)**		**4 (15-20m)**	
	%	LOC	%	LOC	%	LOC	%	LOC
Pre-fire cover litt/moss/duff								
LITT/MOSS/DUFF consumed								
Pre-fire cover shrubs/herbs								
Shrubs/herbs consumed								
Pre-fire tree canopy cover								
Post-fire tree canopy cover								
Pre-fire logs cover								
Pre-fire logs charred								
Component	**2.82 meter radius circle**		**Plot area (20mX5m)**		**15 meter radius circle**		**60 meter radius circle**	
Pre-fire								
Now								
In 50-100 years								
Burn severity: U/VL/L/M/H/VH								

Notes (impacted by soil creep avalanches, snow movement, gofers, ungulates, older fires, bark beetles, etc.)

Seed Source Assessment: Closet 3 seed sources per quad., at least 1 more seed source per quad from before fire tree if closet 3 trees established post-fire. **Observer**______________________



Quad	Spp.	Dist (m)	B/A/S	Slope (deg)	Treetop (deg)	Height (m)
C						
C						
C						
D						
D						
D						
A						
A						
A						
B						
B						
B						

Version 1.4 **09/24/2012** **acansler@uw.edu** **Plot Cover Sheet** **#1**

Fire__________ Plot#__________ Date____________ Recorder____________________

Facilitator Transect (within 0.3 meters of the line):

T.D. start	T.D. end	Type/Spp.	DBH	Decay	B/A	L/Snag Class	LOC
			logs only			trees only	

T.D. = Transect distance (0.05 m); start = when cover starts, end = where cover ends. For Type/Spp. see code sheet. DBH = diameter at breast height of log (0 cm). Decay = Decay class of log; 1 = needles present; 2 = bark abundant; 3 = hard stem; 4 = kick test; 5 = pile. B/A = Log on ground before or after fire. L = bole of live tree, dd; definitions same as overstory assessment. LOC = Level of Certainty for snag class; H (or blank) = high, M = moderate, L = low.

Vegetation Point Intercept (herbs, shrubs, substrates):

X	Substrate	HGT (0.0m)	Species	Species	Species	Species
0.5						
1.0						
1.5						
2.0						
2.5						
3.0						
3.5						
4.0						
4.5						
5.0						
5.5						
6.0						
6.5						
7.0						
7.5						
8.0						
8.5						
9.0						
9.5						
10.0						
10.5						
11.0						
11.5						
12.0						
12.5						
13.0						
13.5						
14.0						
14.5						
15.0						
15.5						
16.0						
16.5						
17.0						
17.5						
18.0						
18.5						
19.0						
19.5						
20.0						

2

Coarse Woody Debris Transect/Vegetation Point Intercept

Plot #____________ Date ______________________ Recorder ___________________________ Fire _____________________

Seedling Assessment:

Y	X end	X start	Species	Height (0.1 m)	1st year	B/A	LOC	B/U	Facilitators Type	Dist	Type	Dist	Type	Dist	Damage Codes		Comments

= Tree number; give same # to trees that share base. **X** = X coordinate at start of section along transect tape (0,2.5,5.0,7.5,10.0,12.5,15.0,17.5). **Y** = - indicates downslope; **+** indicates upslope. **Height** = tree height (0.0m). **1st year** = Y or N. **B/A** = Seeded in before or after fire. **LOC** = Level of Certainty; L = low, M= Medium, H = high. **B/U** = area seedling located burned or unburned by fire. **Facilitators** = facilitators within 0.3 m. **Type** = LOGG (>10cm), BOLD, BEDR, STMP, SNAG, ABLA (or other tree spp. only count tree boles, not branches.) **Dist** = (to nearest 0.05m).

Seedling Assessment   Version 1.3 2012/27/27   acansler@uw.edu   Page ___ of ___ (per plot)   #3

Fire____________ Plot#____________ Date_______________ Recorder_________________________

Overstory Assessment:

Y	X start	X end	Status	Snag class	LOC	Species	DBH	B/A	LOC	B/U	Damage Codes			Comments

#= Tree number. **X** = X coordinate at start of section along transect tape (0,2.5,5.0,7.5,10.0,12.5,15.0,17.5). **Y**= - indicates downslope; + indicates upslope.. **Status** = L = Living; D = Dead; F = Fallen. **Snag class** = only for *dead* and *fallen* trees: RC= Recent; green or brown needles present; FR= Fire; no needles present, evidence of fire but sapwood/heartwood NOT deeply charred; BC = dead before fire; sapwood/heartwood deeply charred; BN = dead before fire; no char; note evidence in comments. **DBH** = diameter at breast height (0.0 cm). **B/A**= did *live* tree establish before or after the fire? **LOC** = level of certainty regarding the snag class or B/A; H = high, M= moderate, L= low B/U = tree burned or unburned by fire

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Fire in the Alpine-Treeline-Ecotone

Plot size 2.82 m

Fire ____________ Plot #_____________ Date______________ Time:_______ Team:____________________
UTM:_____ Easting____________ Northing____________ ± ______ Elevation________ Tree flagging? winds from___°
Soils: Hydric Humus Erode Lithosols Photos__________________
Herb/shrub cover_____ Height______ Major species ____________
Facilitator cover: Trees______Logs______ Bldr______ Bed_______

Component	2.82m radius		15m radius		60m radius	
Pre-fire						
Now						
In 50-100 years						
Burn severity: U/VL/L/M/H/VH						
Severity	%	LOC	%	LOC	%	LOC
Pre-fire cover litt/moss/duff						
LITT/MOSS/DUFF consumed						
Pre-fire cover shrubs/herbs						
Shrubs/herbs consumed						
Pre-fire tree canopy cover						
Post-fire tree canopy cover						
Pre-fire logs cover						
Pre-fire logs charred						

Notes:____

Seedling and Tree Assessment:

Spp.	Height	DBH	Talley	Total

Fire ____________ Plot #_____________ Date______________ Time:_______ Team:____________________
UTM:_____ Easting____________ Northing____________ ± ______ Elevation________ Tree flagging? winds from___°
Soils: Hydric Humus Erode Lithosols Photos__________________
Herb/shrub cover_____ Height______ Major species ____________
Facilitator cover: Trees______Logs______ Bldr______ Bed_______

Component	2.82m radius		15m radius		60m radius	
Pre-fire						
Now						
In 50-100 years						
Burn severity: U/VL/L/M/H/VH						
Severity	%	LOC	%	LOC	%	LOC
Pre-fire cover litt/moss/duff						
LITT/MOSS/DUFF consumed						
Pre-fire cover shrubs/herbs						
Shrubs/herbs consumed						
Pre-fire tree canopy cover						
Post-fire tree canopy cover						
Pre-fire logs cover						
Pre-fire logs charred						

Notes (impacted by soil creep avalanches, snow movement, gofers, ungulates, older fires, bark beetles, etc.) ____________________

Seedling and Tree Assessment:

Spp.	Height	DBH	Talley	Total

Height classes (m): 0-0.05, 0.05-0.1, 0.1-0.4, 0.4-0.9, 0.9-1.4 DBH classes (cm): <1, 1-2.5, 2.5-5, 5-10, 10-15, 15-20, 20-30, 30-40
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