Sustainability of Historic Wildfire Refugia in Contemporary Wildfire Events

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Abstract

In fire-adapted forest ecosystems, spatial heterogeneity of fire effects is essential to maintaining overall species and habitat diversity across the landscape. Forest patches that are minimally-affected by wildfire without transitioning into a different successional state (termed ‘refugia’) maintain critical habitat for fire-sensitive species. Due to fire suppression and climate change, historically-persistent wildfire refugia may be vulnerable to loss. We investigated historically-persistent wildfire refugia and surrounding non-refugial matrix classified by Camp et al. (1997) after fires burned through the original study area in 2012 under extreme weather conditions. We found that previously classified historic wildfire refugia experienced greater fire effects than the non-refugial matrix, yet the majority of sampled forest stands persisted in the pre-fire successional state. This result demonstrates that individual wildfire refugia may not be persistent through time indefinitely, but that some patches persist as refugia within a fire area even during extreme wildfires.

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CHAPTER 1: Sustainability of Historic Wildfire Refugia in Contemporary Wildfire Events

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1. Introduction

Wildfire is an integral natural ecosystem process in the ecology of many forests worldwide, including the conifer forests of the inland northwest United States (Agee 1993). In these fire-adapted ecosystems, wildfire is a primary influence upon the spatial heterogeneity of successional forest patches and is a driver of both forest composition and structure (Stine et al. *in press*). Though wildfire is a primary ecological process in inland northwest forests, not every stand is affected equally by fire (Lentile et al. 2007) and the differential degrees of ecological change due to fire is known as burn severity (Morgan et al. 2001). The spatial heterogeneity of fire effects and burn severity are important to the ecological functioning and trajectory of fire-adapted forests as a whole (Agee 1993). For example, severe wildfire impacts (e.g., stand replacement fire and high tree mortality) have occurred naturally for millennia and provide an ecological backdrop for fire specialist species (Hutto 2008). However, less studied but equally important ecologically are forest patches that remain unburned or experience a low degree of change from a wildfire event; these patches are known as ‘wildfire refugia’ (Kolden et al. 2015).

Defined broadly by Keppel et al. (2012), refugia are landscape patches where components of biodiversity are able to retreat, persist, and potentially expand under changing environmental conditions. More specifically for wildfire, refugia are landscape patches in which the pre-fire successional state of the patch persists after a fire occurs due to little ecological change to the patch (Camp et al. 1997, Kolden et al. 2012, Kolden et al. 2015). In previous studies, areas within fire perimeters that do not experience fire effects at all (e.g., Delong and Kessler 2000, Schwilk and Keeley 2006), or that experience fire effects at a lower severity than surrounding areas (e.g., Camp et al. 1997, Turner et al. 1999) have been treated as wildfire refugia. In this study, we adhere to Keppel et al.'s definition of refugia by classifying wildfire refugia as forest patches that persist in the pre-fire successional state after the fire event. Characteristics of what constitutes a wildfire refugia vary by organism and ecosystem as well, with previous wildfire refugia research encompassing taxon as diverse as invertebrates (Ghandi et al. 2001, Swengel and Swengel 2006, Brennan et al. 2011), vertebrates (Gaines et al. 1997, Robinson et al. 2013), non-woody plants (Pfab and Witkowski 2000, Hylander and Johnson 2010), trees (Schwilk and Keeley 2006), and soil fauna (Zaitsev et al. 2014). In this study we elected not to focus on refugia for a single species, but to instead study wildfire refugia more broadly as habitat patches that remain suitable for fire-intolerant species. Identifying the location and formation of wildfire refugia is important, as wildfire refugia in fire-adapted ecosystems maintain species diversity due to critical habitat being retained in refugial patches (Agee 1993). Additionally, wildfire refugia play a critical role in the post-fire trajectory of proximal landscape patches by providing proximate seed sources and vegetative propagules for post-fire vegetative regeneration (Halpern 1988, Turner et al. 1999).

The occurrence of wildfire refugia depends upon a number of factors that vary spatially and temporally and also have the ability to confound each other. These factors comprise the traditional 'fire behavior triangle' of fuels (i.e., live and dead vegetation), topography, and weather (Countryman 1966). At the meso-scale, forest composition and structure are influenced by topography, weather, soils, geomorphology, and processes such as fire (Stine et al. *in press*). These factors interact together to create spatial heterogeneity in vegetation and forest structure, which is primarily maintained by fire events in fire-adapted ecosystems like the inland northwest (Martin and Sapsis 1991, Bowman et al. 2009). For example, vegetation characteristics such as stand age or structure can either increase or decrease the likelihood of fire occurrence (Oliver and Larson 1990, Kushla and Ripple 1997, Alexander et al. 2006), or can minimize or exacerbate the effects when a fire does occur (Turner et al. 1989). Topographic complexity influences burn severity (Dillon et al. 2011, Cansler and McKenzie 2014) and refugia formation as well (Camp et al. 1997). Generally, bottom-up factors such as vegetation and topography exert a greater influence on burn severity than top-down controls such as weather or climate (Dillon et al. 2011, Birch et al. 2015), but in extreme fire weather events, local weather conditions may override vegetative and landscape effects on burn severity altogether (Bessie and Johnson 1995, Haire et al. *in review*). Furthermore, human activities such as stand management have the potential to influence the severity of burn patterns and the formation of refugial patches (Prichard and Kennedy 2014).

The capacity of an ecological system to absorb changes induced by ecosystem processes (including fire) such that the ecosystem retains essentially the same function, structure, and feedbacks is a concept known as ecological resilience (Holling 1973, Beisner et al. 2003, Walker et al. 2004). Similar to the concept of wildfire refugia as being persistent after a wildfire, ecological resilience is the magnitude of perturbation from ecosystem processes that can be absorbed by an ecosystem before the ecosystem transitions into a different state (Holling 1996). At the ecosystem scale, an ecosystem that experiences a pattern of fire effects similar to the historical fire effects typical for that ecosystem exhibits a high degree of ecological resilience. However, even though an ecosystem as a whole may exhibit a high degree of ecological resilience, individual patches within the ecosystem may experience differential degrees of ecological change (Burton et al. 2008); this is especially true for forests with a historically mixed-severity fire regime like the inland northwest (Agee 2005). Ecosystems with mixed-severity fire regimes exhibit a multiplicity of post-fire successional states and ecological transitions due to fire (Agee 2005), the interaction of which can be conceptually framed using state-and-transition models (e.g., Keane et al. 2001, Keane and Karau 2010, Haugo et al. 2015).

In order to assess ecological resilience at the ecosystem scale, it is imperative to understand the boundaries of historical ecosystem components and processes, which is a concept known as the historical range of variability (HRV; Morgan et al. 1994, Keane et al. 2009). The historical range of variability of successional states and ecological transitions provides a useful frame of reference for determining the extent of ecological change that can occur in an ecosystem while still being within the bounds of historic states and transitions (Morgan et al. 1994, Keane et al. 2009). When ecological impacts from wildfire exceed the bounds of the HRV for the fire regime, an ecosystem may exhibit reduced resilience to the natural process of fire (Keane et al. 2009). Such loss of ecological resilience may potentially lead to the transformation of the ecosystem to a different ecological regime altogether, especially in an era of global environmental change (Walker et al. 2004). Thus, analyzing fire effects and refugial occurrence within the context of the historical range of variability for successional states and ecological transitions provides an assessment of contemporary ecological resilience.

There is evidence that environmental change may reduce the resilience of wildfire refugia when compared to historical conditions, and such loss of ecological resilience is a contributor to loss of biodiversity and ecosystem services (Mackey et al. 2002, Folke et al. 2004, Keppel et al. 2012, Robinson et al. 2013). For example, local alterations in fire regimes have greatly changed both fire frequency and intensity from historical normals in inland northwest forests (Hessburg and Agee 2003). Euro-American settlement and associated timber harvesting, along with a century of fire exclusion, have caused a decline in open early-seral stand structures and species while simultaneously increasing multilayered late-seral forests with elevated fuel loading and fire intolerant species (Hessburg et al. 2000). Subsequently, these altered forest structures have affected the spatial distribution and intensity of wildfires, increasing the risk of stand-replacing fire in particular (Hessburg et al. 2000). Another primary contributor of environmental change is the alteration of historical climate parameters. Inland northwest forests are already experiencing conditions conducive to increased occurrence and duration of wildfires, and these trends are projected to continue through the 21st century (Pierce et al. 2004, Littell et al. 2009, Abatzoglou and Kolden 2013). Increases in climate-induced drought and heat stress in vegetation have already shown to increase tree mortality due to fire in western US forests (van Mantgem et al. 2013). Furthermore, recent studies have shown that the proportion of areas burned in wildfires at high severity has been increasing for some ecosystems (Miller et al. 2008, Miller and Safford 2012). If the proportion of high severity burned areas is increasing across the broader western United States as conjectured by Miller et al. (2008), then the proportion of areas burned at all other severities must consequently be decreasing. One potential manifestation of this decrease is that the amount of unburned and lower severity burned areas such as wildfire refugia could be decreasing in contemporary wildfires.

In contrast to studies suggesting the proportion of area burned at high severity is increasing, Kolden et al. (2012, 2015) found no trends in proportion of unburned and persistent patches within fire perimeters. However, both Kolden et al. analyses, as well as the Miller et al. (2008) and Miller and Safford (2012) studies, were limited by the use of remotely sensed data to quantify trends in fire severity, and by the short temporal duration of their studies (1984-present). Other studies that have characterized the spatial occurrence of wildfire refugia with field-based techniques have primarily done so through post-hoc classifications of refugial versus non-refugial areas within a single wildfire event (e.g., Eberhart and Woodward 1987, Kushla and Ripple 1997, DeLong and Kessler 2000, Alexander et al. 2006, Román-Cuesta et al. 2009, and Larson and Churchill 2012 *and sources therein*), but such studies do not explore the persistence of wildfire refugia through multiple historic wildfire events.

To our knowledge, no field-based studies have specifically assessed the persistence and resilience of wildfire refugia based on the pre-fire identification of long-term wildfire refugia, with the exception of one study conducted by Camp et al. (1997) in central Washington State, USA, which addressed the historical persistence of wildfire refugia and provides a base for assessing refugial persistence in a changing environment. Camp et al. examined forest structure, tree age, and species composition of stands within the Wenatchee National Forest, classifying historic wildfire refugia as forest patches that had been minimally affected by fire events for at least 140 years while the surrounding forest matrix had experienced greater fire effects in those same fire events. In 2012, two wildfires occurred in Camp et al.'s study area, providing an opportunity to assess contemporary fire effects on the original sampled sites. Opportunistic field re-sampling allowed for an evaluation of both Camp et al.'s pre-fire refugial classification and for examining the persistence of historic wildfire refugia under changing fire regimes and climatic conditions.

The primary objective of this study was to determine if the 2012 wildfires produced contemporary distributions of wildfire refugia and ecological change that is comparable to the historical range of variability for refugia distributions and fire-induced ecological transitions. We addressed this question by (1) quantifying and comparing fire effects between refugial and non-refugial plots as classified pre-fire by Camp et al. (1997), (2) comparing distributions of post-fire successional states and ecological transitions to historic distributions as provided in the literature addressing the historical range of variability, and (3) investigating the relative contributions of vegetation, topography, and fire weather to both post-fire successional states and fire-induced transitions of successional states. More broadly, this research evaluates how the occurrence of wildfire refugia and distribution of fire effects may be altered from the historical range of variability by environmental change.

2. Materials and Methods

2.1 Study Area

The study site is located in the Swauk Late Successional Reserve (LSR) of the Wenatchee National Forest (NF) of central Washington State, USA, the same location of the original Camp et al. (1997) study sites and of the 2012 fires of interest (Figure 1). Late Successional Reserves were created through the Northwest Forest Plan to protect and enhance the condition of late-successional and old-growth forest ecosystems, and as such permit only limited stand management in the reserve (USDA and USDOI 1994). Prior to attaining LSR status in 1995, this area was subject to more intensive management, including selective timber harvesting, clear-cutting, road building, and mining, particularly in lower drainages (Camp et al. 1997).

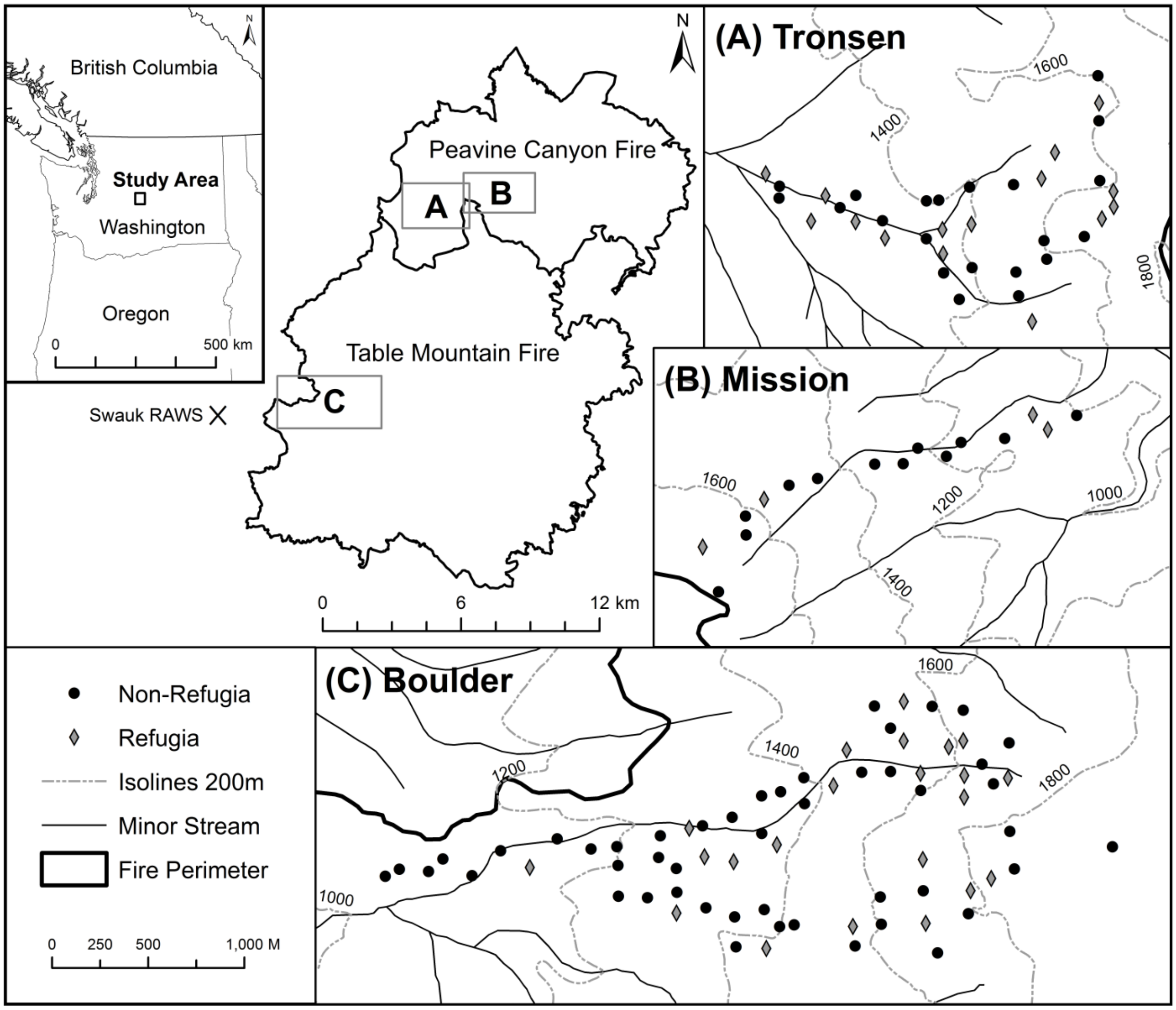

Figure 1: Study area in the east Cascades of central Washington State, USA, with burn perimeters of the 2012 Peavine Canyon and Table Mountain fires and spatial locations of the Camp et al. (1997) classified refugial and non-refugial plots in the three study drainages (Tronsen, Mission, and Boulder).

The study area is located at the extreme eastern edge of the Cascade mountain range, extending into the dry interior Columbia River Plateau to the east. Bedrock in this area consists primarily of assorted sedimentary rocks types with outcroppings of younger basalts (Alt and Hyndman 1994). Typical landforms in the area inlcude steep scarp slopes, hogback ridges, basalt plateaux, talus, scree slopes, and earthflows (Camp et al. 1997). Extensive historical mountain glaciation created steep, long slopes in the drainages which are mainatained by active erosion and downcutting (Williams and Smith 1990). Sampled plots in this study ranged in elevation from 1027 meters to 1912 meters.

Vegetative communities in the Swauk LSR form a heterogeneous landscape due to strong responses to the dissected topography, precipitation gradient, and insolation differences (Williams and Smith 1990). At more xeric sites, lower elevations, and south facing aspects, open-canopy ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Psuedotsuga menziesii*) stands are common. At more mesic sites, higher elevations, and north-facing slopes, stands of subalpine fir (*Abies lasiocarpa*) and lodgepole pine (*Pinus contorta*) are more typical. However, specific site conditions can cause immediate juxtaposition of disjunct forest stands. Table 1 details relative abundance and species diversity of trees sampled in this study, revealing that grand fir (*Abies grandis*) and Douglas-fir were the most common trees encountered. Open meadows and barren rockfields are also common but were not sampled in this study.

Table 1: Species composition of surveyed trees from study plots.

Species	Count	Percent
Abies grandis	481	39
Abies lasiocarpa	138	11
Larix occidentalis	44	4
Picea engelmannii	13	1
Pinus contorta	53	4
Pinus ponderosa	77	6
Populus spp.	29	2
Pseudotsuga menziesii	378	31
Other/Unknown	7	1
Total	1,220	100

Wright and Agee (2004) found that immediately outside the Swauk LSR the historical fire regime varied spatially by vegetative type, with a mean fire return interval (MFRI) of seven to 43 years, and with large fires occurring every 27 years. Historical fire severity varied in the study area as well, with drier forest types experiencing low fire severity while more mesic forest types experienced occasional moderate and high severity fires (Wright and Agee 2004). However, fire regimes have been significantly altered from their historic range of variability in the inland northwest. Fire frequency declined dramatically around 1900, coinciding with the start of commercial logging (Wright and Agee 2004) and the advent of active fire suppression in the Wenatchee NF (Holstine 1992). Consequently, Everett et al. (1992) found no evidence of fire in the Swauk LSR from 1900 to 1990.

2.2 The 2012 Fires

The Table Mountain and Peavine Canyon (as part of the Wenatchee Complex) fires burned simultaneously after igniting from lightning strikes in early September 2012. These fires eventually merged, creating a unified burn area of 25,274 ha. Both fires were extinguished in mid-October through a combination of suppression efforts and precipitation (NIFC 2012). These fires burned 226 sample sites of the Camp et al. (1997) study in three different drainages, where 43 and 183 plots were classified by Camp et al. as refugial and non-refugial respectively.

The fires burned under anomalously dry and warm weather conditions compared to 1985 to 2014 climate data recorded at the Swauk Remote Automatic Weather Station (RAWS) approximately 2 kilometers west of the fire perimeter (WRCC 2014). Average air temperature for the July through September fire season was higher than normal (83^{rd} percentile), and average air temperature for September when the fires ignited was much higher than normal (93^{rd} percentile). Average relative humidity for the fire season was lower than normal (35^{th} percentile), and average relative humidity for September was much lower than normal (3^{rd} percentile). Average wind speed for the fire season was slightly lower than normal (36th percentile), but wind gusts were much higher than normal (86th percentile). Precipitation was slightly below normal for the fire season (38th percentile), but late-season drought was particularly pertinent as the study area went 52 consecutive days without measurable precipitation prior to fire ignition and 34 days without afterwards. Such fire weather conditions have the ability to produce extreme fire behavior in this type of environment (Schroeder et al. 1964).

2.3 Data

2.3.1 Field Data

Field data was collected in summer and fall 2014 by resampling the locations of the Camp et al. (1997) sample sites that burned in 2012 (Figure 1). Since the Camp et al. plots were not permanently monumented and prohibitively difficult to relocate in-field with high accuracy, new plots were established as close as possible to the original plots and within the same forest stand. The original Camp et al. (1997) plots were established using topographic maps, hip chain, and sighting compass in the field, with the final plot locations being annotated on topographic maps. In order to determine GPS coordinates of these plots, the annotated topographic maps were first digitized into a Geographic Information System (GIS) database and then plot coordinates were navigated to in-field with a handheld GPS unit. Time and resources did not allow a revisit to every Camp et al. plot within the fire area, so a transect of random plots was sampled from the bottom to the top of each study drainage in order to ensure sampling of the entire elevational gradient in each drainage. GPS locations that fell in barren areas or in unsafe field sites (e.g., cliff edges or steep stream embankments) were relocated to the nearest suitable forested site. Ultimately we revisited 41 refugial sites and 81 non-refugial sites for a total of 122 sample sites across the three study drainages (Appendix A).

At each sample site, a modified replication of the Camp et al. (1997) study protocol was conducted (Appendix B). Plot center was monumented at the ascribed GPS coordinates and a 15.2 meter diameter circle was circumscribed. Variables collected at the plot level at each site included four topographic, nine vegetative, and four fire effects variables (Table 2). The topographic variable of plot aspect was measured in-field using degrees, but was decomposed into northness and eastness indices for analysis. We also collected variables at the tree level for each plot, where using a sweeping transect from an azimuth of 0^{o} (north) we sampled the first ten trees in the plot with a diameter at breast height (DBH) greater than 12.6 centimeters. We tagged and assessed these trees for three demographic and five fire effects variables (Table 3). Three additional vegetative variables at the plot level (average DBH, maximum DBH, and pre-fire plot basal area) were derived from the DBH measurements of the 10 sampled trees on each plot (Table 2). A total of 1,220 trees were surveyed for this study.

Comprehensive burn severity was also assessed at each site using the Composite Burn Index (CBI) protocol developed by Key and Benson (2006). We chose to use CBI as the integrative plot level burn severity metric because it is commonly used to assess burn severity across landscapes and for field validation of remotely sensed burn severity. Due to limitations in the CBI protocol itself (Morgan et al. 2014), we also chose to assess burn severity for the five different ecologically-based metrics at the tree level as described in the preceding paragraph. The area of CBI analysis for this study was modified to be a 30 meter by 30 meter square to correspond to the size of a Landsat pixel; additionally, the plot was oriented in the cardinal directions to align with the Camp et al. (1997) plot azimuths. While CBI is normally conducted one year post-fire (Key and Benson 2006), we were unable to conduct field work in 2013. However, CBI has been previously utilized to assess fire effects two years post-fire (Zhu et al. 2006); doing so allows an assessment of longer-term fire effects while also capturing delayed mortality in the tree strata.

Table 2: Plot topographic, vegetative, and fire effects variables collected in-field

Data Type	Variable	Definition
Topographic	Aspect Northness	cos(π/2-aspect); range from -1 (south) to 1 (north)
	Aspect Eastness	sin(π/2-aspect); range from -1 (west) to 1 (east)
	Slope	Degrees; measured by clinometer
	Elevation	Meters; measured by GPS
	Topography Type	Ten option categorical classification (Appendix C)
Vegetative	Max Canopy Height	Meters; measured by Impulse Laser
	Species Present	ABGR, ABLA, LAOC, PICO, PIEN, PIPO, POTR, PSME
	Canopy Structure	Presence/absence of overstory/subcanopy strata
	All Trees Pre-fire	Count of all trees alive in plot pre-fire
	Overstory Trees Pre-fire	Count of all large trees alive in overstory strata pre-fire
	Subcanopy Trees Pre-fire	Count of all trees ≥12.6 cm DBH alive in subcanopy strata pre-fire
	Total Canopy Cover Pre-fire	Ocular estimate of pre-fire canopy cover for all tree strata
	Overstory Canopy Cover Pre-fire	Ocular estimate of pre-fire canopy cover for overstory strata
	Subcanopy Canopy Cover Pre-fire	Ocular estimate of pre-fire canopy cover for subcanopy strata
	Average DBH	Average DBH of 10 sampled trees on plot
	Maximum DBH	Maximum DBH of 10 sampled trees on plot
	Pre-fire Plot Basal Area	Average basal area of 10 sampled trees on plot [$\pi(DBH/2)^2$] * count of trees on plot. Unit: m^2 basal area / 900 m^2 plot area
Fire Effects	Total Tree Mortality	Count of all tree mortality in plot post-fire
	Overstory Tree Mortality	Count of all large tree mortality in overstory strata post-fire
	Subcanopy Tree Mortality	Count of all subcanopy tree mortality in subcanopy strata post-fire
	Total Plot CBI	Composite Burn Index protocol (score from 0 to 3)

Table 3: Tree demographic and fire effects variables collected in-field

Data Type	Variable	Definition
Demographic	Species	Field Identification
	DBH	Field measure; cm
	Secondary Stress	Presence of: Fire, Freezing, Fungus, Insect, Mechanical, Mistletoe, Rot
Fire Effects	Mortality	Fire-induced tree death
	Percent Bole Char	Maximum percent of basal bole with visible char
	Bole Char Max Height	Maximum height of continuous char on bole (m)
	Percent Foliage Scorch	Ocular estimate of pre-fire living foliage scorched or girdled
	Percent Foliage Torch	Ocular estimate of pre-fire living foliage torched by fire

2.3.2 Data Quality Assurance between Resampled and Original Field Plots

To ensure that plots sampled in 2014 did not significantly differ from the plots which Camp et al. (1997) originally sampled and classified into potential refugia, we conducted a paired-plot assessment using the original plot data and stand delineations produced from Camp et al. (1997). First, stand delineations and aerial imagery were used to determine if original and resampled plots fell in the same forest stand. If the original and resampled plot pair was not visually within the same stand, then topographic and vegetative attributes of each plot were compared for similarities. Thirteen sampled plots were not able to be confidently matched through this qualitative comparison and were excluded from further paired-plot analysis which resulted in a total of 109 confident paired-plot matches. We then used a paired t-test with Welch modification for non-normality to test for differences in the topographic setting of both sample sets (Welch 1947). Elevation was the only topographic attribute that was statistically different between the two sample years ($p<0.001$); however, the difference of 14 meters in elevation is not considered ecologically significant over the range of our study area and is also likely due to sampling methodology differences between sample years (altimeter versus handheld GPS measure). Using these 109 matched plots instead of the entire sampled plot population provides a conservative match to the Camp et al. (1997) plot-level data.

Sampled trees from the Camp et al. (1997) study and this study were compared using the 109 paired plot matches discussed previously. The distribution of sampled tree species was determined to be statistically different ($p<0.001$) by a chi-square test of independence. Likewise, the DBH of resampled trees was statistically smaller ($p<0.001$) than the original trees as determined through a t-test with Welch modification. Differences in species and DBH of trees sampled are likely due to differences in protocol between the sample years: the number of trees Camp et al. sampled per plot varied from one to seventeen, stumps were included in the survey, and the largest trees in each plot were intentionally sampled. These statistical differences and differences in sampling protocol make the comparison of data at the tree level untenable.

2.3.3 Fire Weather Data

Fire weather was broadly summarized by progression intervals and hourly weather recordings from the Swauk Remote Automatic Weather Station (RAWS). Although such characterization of fire weather behavior is relatively coarse in scale compared to the other data collected in this study, Collins et al. (2007) and Prichard and Kennedy (2014) both reported significant relationships between weather assigned by progression intervals and burn severity using an identical protocol. Progression intervals derived from daily infrared active fire detection flights were downloaded from the National Interagency Fire Center's (NIFC) incident specific database (NIFC 2012). Fire progression was mapped daily throughout the duration of the fire, except for days when incident crews were unable to fly infrared detection. Because infrared detection flights typically occurred late at night or in early morning, all intervals included periods of both daytime and nighttime burning.

We analyzed the weather variables of maximum temperature, minimum relative humidity, maximum wind speed, and average wind speed for each progression interval due to the effect these variables impart on fire behavior (Agee 1993) and their use in previous studies (Pritchard and Kennedy 2014). Hourly data were downloaded for the Swauk RAWS (wrcc.dri.edu) and then assigned to the correct progression interval. Values for each variable were then computed by taking either the maximum hourly value (maximum temperature, maximum wind speed), minimum hourly value (minimum relative humidity), or average of all hourly values (average wind speed) for the respective progression interval. Because all plots that burned during the same progression interval inevitably had identical weather values due to only one RAWS station being available, this fire weather variable assignment methodology resulted in 14 unique progression intervals for analysis.

2.4 Statistical Analysis

2.4.1 Comparison of Fire Effects between Resampled and Original Field Plots

We used the 109 plots determined to be confident matches from the data quality assurance step (section 2.3.2) and tested for differences in burn occurrence and burn severity between refugial (n = 36) and non-refugial (n = 73) plots. Under Camp et al.'s refugial definition, refugial plots would have been only 'minimally affected' during a fire event. Because Camp et al. did not develop quantitative criteria for what constituted a plot being 'minimally affected' by fire, we initially used a conservative assumption that refugial plots do not experience any fire effects in a fire event. Under this assumption, fire would have been expected to not occur in the Camp et al. classified refugial plots, resulting in an expected proportion of 33 percent unburned plots (36 of 109 plots). Differences in burn occurrence between refugial and non-refugial plots were assessed with a chi-square test of independence. Preliminary results revealed that of the 109 paired plots, only six of these did not experience fire effects in 2012, and that only one of these six unburned plots was classified as refugial by Camp et al. (1997), though this difference in burn occurrence was not statistically significant (chi-square test of independence, $\chi^2 = 0.768$, $p = 0.381$).

Based on the small number of unburned plots and our field observations that some plots that burned did not experience a high degree of ecological change due to fire (i.e., fire only 'minimally affected' the plot), we posited that our conservative assumption of refugial plots remaining unburned in a fire event was too restrictive for a definition of refugia. Subsequently, we developed a secondary hypothesis about what qualified as refugial based on state-and-transition theory, where a plot was considered refugial if it persisted in the pre-fire state after the 2012 fire event (Keppel et al. 2012). Under this secondary hypothesis, we were interested in assessing differences in the degree of ecological change due to fire between refugial and non-refugial plots which we assessed with a Wilcoxon-signed-rank test due to the non-normal distribution of the data. The Wilcoxon-signed-rank test results in a W-value where lower W-values correspond with lower p-values. Comprehensive burn severity at the plot level was represented by the total plot CBI metric. More ecologically specific differences in fire effects included maximum bole char height, percent bole charred, percent foliage scorched, percent foliage torched and percent tree mortality of overstory and subcanopy tree strata.

2.4.2 Classification of Sampled Plots into Successional States

The vegetative characteristics of different successional states within a forested ecosystem depend on the vegetative community of the ecosystem itself (Shugart 1984). Thus, in order to assign successional states to our sample plots, we first needed to assign the type of forested ecosystem in which each plot occurred. A nationwide vegetative community classification called the Biophysical Setting (BpS; NatureServe) was developed as part of the LANDFIRE resource management and planning tool (Rollins 2009). Biophysical setting and other LANDFIRE vegetation products are widely used in state-and-transition studies (Keane et al. 2001, Strand et al. 2009, Keane and Karau 2010, Haugo et al. 2015). The 122 plots sampled in 2014 were assigned to one of the three most common Biophysical Settings located in the study area through analysis of the 2014 quantitative plot data, recorded qualitative field observations, and photos for each plot. Since each BpS models covers a broad geographical area and cannot account for more localized variation within a single model, BpS models were refined with locally available information on the habitat types of the Wenatchee National Forest (Lillybridge et al. 1995; Table 4). Forest types will hereafter be referred to by the Wenatchee NF correlate name (i.e., Douglas-fir, Grand fir, or Subalpine fir series).

Table 4: Ecosystem Distribution by BpS Model and Wenatchee NF Correlate

BpS Model Name	Wenatchee NF Correlate (Lillybridge et al. 1995)	Plots
Northern Rocky Mountain Dry-Mesic Montane Mixed Conifer Forest	Douglas-fir Series	17
East Cascades Mesic Montane Mixed-Conifer Forest and Woodland	Grand Fir Series	84
Rocky Mountain Subalpine Dry-Mesic Spruce-Fir Forest and Woodland	Subalpine Fir Series	13

Once a BpS model was assigned to each plot, pre- and post-fire successional states were assigned according to the Vegetation Dynamics Development Tool (VDDT; ESSA Technologies Ltd. 2007) models for each respective BpS setting (Rollins 2009; Table 5; *see Appendix D for proportions within each BpS model*). The VDDT model for each of the three BpS settings uses five distinct successional/structural classes (hereafter, referred to as “successional states”): early development, mid-development closed canopy, mid-development open canopy, late development open canopy, and late development closed canopy. Within each BpS model there are different attribute criteria for what constitutes a particular successional state. Quantitative and qualitative criteria used to assign successional state to each plot are summarized in Table 6. Post-fire successional state was classified through post-fire vegetation observed during the 2014 field season. For pre-fire successional state, in-field estimates of canopy cover and counts of living trees pre-fire were used as best approximations for pre-fire vegetative structure and composition. Field notes and plot photos were used to refine this classification when quantitative data alone proved inconclusive.

Table 5: Distribution of VDDT assigned successional states for all 122 sampled plots according to pre-fire and post-fire categorization for all BpS models.

Successional State	Pre-fire State		Post-fire State	
Early Development	0	0%	23	19%
Mid-Development Closed Canopy	47	39%	12	10%
Mid-Development Open Canopy	39	32%	55	45%
Late Development Open Canopy	11	9%	24	20%
Late Development Closed Canopy	25	20%	8	6%

Table 6: Quantitative and qualitative criteria used to classify plots into pre- and post-fire successional states

Quantitative	Species Present
	Canopy Cover
	Canopy Height
	Tree DBH
Qualitative	Tree Relative Canopy Position
	Fuel Model

Once pre-fire and post-fire successional states were assigned to each plot, the transition from one successional state to another due to fire effects was assessed. There were three distinct transitions a plot could have taken due to fire effects (hereafter referred to as “ecological transitions”: (1) plot was maintained in current successional state, (2) plot canopy was thinned from closed to open canopy structure of the same development stage, or (3) plot transitioned to early development successional state. Half (50%) of the sampled plots did not change successional state due to fire effects, while 31 percent of sampled plots transitioned from closed to open canopy structure (Table 7). Only 19 percent of sampled plots transitioned to early development successional state. A conceptual diagram of the role of fire severity in controlling processes and pathways of ecological transitions for this site is visually summarized in Figure 2, modified from Kolden et al. (2015).

Table 7: Post-fire ecological transitions of successional states.

Ecological Transition	Plots	Percent
Maintained Successional State	61	50%
Canopy Thinned from Closed to Open	38	31%
Transitioned to Early Successional State	23	19%

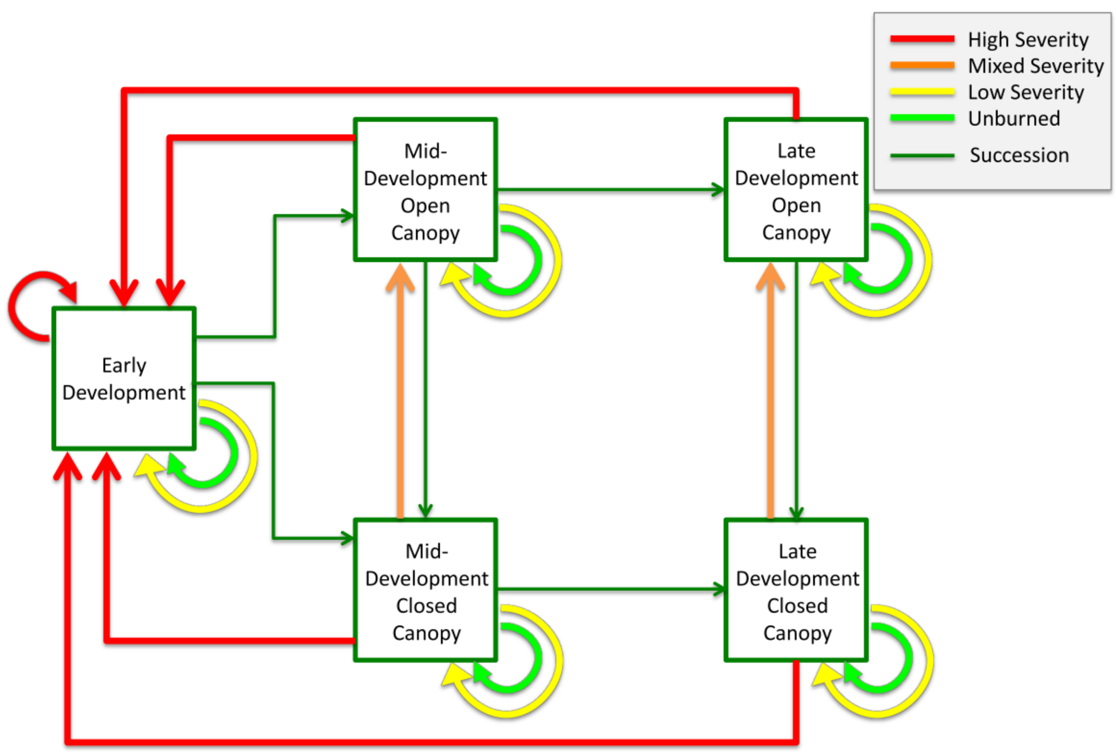

Figure 2: State-and-transition model of VDDT successional states and ecological transitions as affected by differential wildfire severity (modified from Kolden et al. 2015).

2.4.3 Predictors of Ecological Transitions and Post-fire Successional States

We used classification and regression tree (CART) analysis to determine which predictor variables had the greatest influence on successional states and ecological transitions, opting for classification trees in order to group our data into discrete response categories (Breiman et al. 1984). CART analysis is ideal for complex ecological datasets because CART models are non-parametric, can include both continuous and categorical variables, can handle missing data (De'ath and Fabricius 2000) and can reliably be used with spatially-dependent data (Bel et al. 2009). Additionally, CART analysis more aptly fits higher-order interactions and strong non-linearities in predictor variables that can be missed in other analyses such as regression (De'ath and Fabricius 2000), which makes it an ideal methodology to elucidate causal factors of fire-induced ecological transitions. To build a tree, CART analysis employs a recursive partitioning algorithm that iteratively partitions the dataset into binary groups such that within-group homogeneity is maximized until further splits of the data fail to improve the fit of the overall model (Breiman et al. 1984). The resulting trees built by CART analysis are represented graphically, are transparent, and are easy to interpret (De'ath and Fabricius 2000).

Due to low numbers of some successional states under the five successional state categories used in VDDT models, we regrouped the five successional states into three classes based on development structure of the plot (e.g., mid-successional open canopy and mid-successional closed canopy successional states were both reassigned to mid-development stage). This reclassification resulted in three development stage groups of early, mid-, and late development (hereafter referred to as "development stages"; Table 8). Reducing the five VDDT successional states into three development stages not only simplifies interpretation of resulting classification trees, but it also reduces the potential for overfitting our classification, which is a recognized drawback to using categorical response variables with small numbers in each class (Breiman et al. 1984). This regrouping is justifiable because canopy closure is independent of development stage and because the difference between open and closed canopy successional states was based on one variable (canopy cover) while development stage was based on multiple variables.

Table 8: Distribution of sampled plots post-fire grouped by development stage.

Group	Plots	Percent
Early	23	19%
Mid-	67	55%
Late	32	26%

We then created two classification trees, with the response variables being (1) ecological transition due to fire, which provides an indicator of how fire influences the successional trajectory of the forest, and (2) development stage post-fire, which provides a measure of the vegetative structure of the landscape immediately post-fire. For the response variables in both classification trees, we identified 39 potential predictor variables to include in the initial models, based on the broader categories of topography (6 variables), vegetation (20 variables), fire weather (5 variables), and fire effects (8 variables). Of these potential predictor variables, 26 were quantitative and the remaining 13 were qualitative (Appendix E). Fire effects variables were included in these initial models to determine if fire severity as a whole was a more suitable predictor of ecological transitions or post-fire development stage in comparison to the individual components of the fire behavior triangle.

In order to eliminate highly co-varied quantitative variables from classification tree analysis, a covariance matrix was created where variables were eliminated if Spearman's Rho exceeded 0.75 (Appendix F*; see protocol in Birch et al. 2015*). If a plot level variable co-varied strongly with the sub-plot level variable of the same metric, the sub-plot level variable was removed first (5 variables removed). Next, variables highly co-varied with more than one other variable were removed (2 variables removed). Finally, when a variable highly co-varied with only one other variable, the decision of which variable to eliminate was based on which variable we had more confident measures of for vegetative variables or upon literature review for weather variables (5 variables removed). Ultimately, 12 variables were eliminated from analysis due to high covariance with other predictor variables, leaving 27 total potential predictor variables for inclusion in classification tree analysis, 14 of these being quantitative and 13 being qualitative (Appendix E). Of these remaining variables, six were topographic, 15 were vegetative, three were fire weather, and three were fire effects.

Using the 27 remaining independent predictor variables, we constructed two classification trees in order to determine the best indicators of (1) fire-induced ecological transitions and (2) post-fire development stages. We used the rpart function in the R statistical language (R Core Team 2014) to create our initial models using a minimum node size of ten to attempt a further split, a minimum terminal node size of three, and a reduction of error criteria of 0.001. These initial models included a high number of nodes which overfit the data, so both resulting trees were pruned to the optimal number of nodes based on the minimization of cross validation error (De'ath and Fabricius 2000).

3. Results

3.1 Preliminary Assessment of Camp et al. (1997) Refugial and Non-Refugial Plots

Comparison of fire effects between Camp et al. (1997) refugial and non-refugial plots revealed a trend in differences in burn severity, where classified refugial plots experienced greater fire effects (Figure 3). Refugial plots had higher comprehensive burn severity scores as assessed through the total plot CBI metric ($W = 1015.5$, $p = 0.0582$). For the tree-level severity metrics, percent total tree mortality ($W = 921.5$, $p = 0.0196$), percent understory tree mortality (W = 867, p = 0.0116), average bole char height (W = 1042, p = 0.0802), and average foliage scorched (W = 899, p = 0.00754) were all statistically higher for refugial plots. Percent overstory tree mortality and percent bole char were also higher for refugial plots, but this difference was not statistically significant. Foliage torched showed no significant difference between refugial and non-refugial plots, but this result is likely attributable to low overall levels of foliar torching with the exception of a few highly torched plots resulting from crown fires.

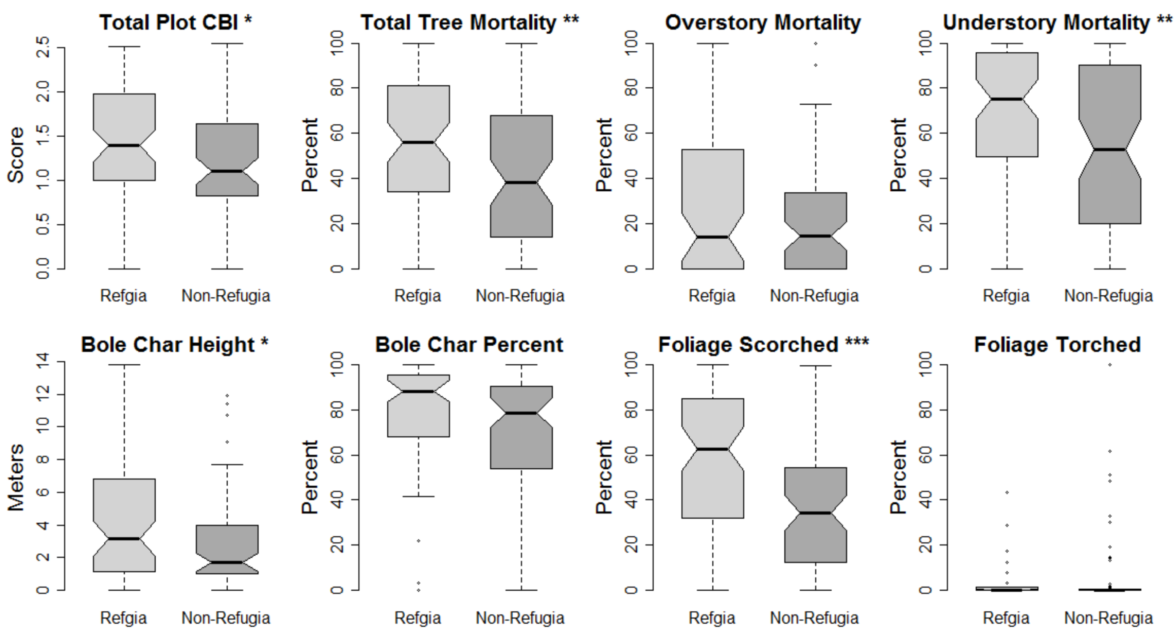

Figure 3: Comparison of fire effects between Camp et al. (1997) refugial and non-refugial plots for eight different burn severity metrics. *p ≤ 0.10, **p ≤ 0.05, ***p ≤ 0.01.

3.2 Post-fire Successional States

Classification into successional states and development stages revealed a heterogeneous landscape both pre- and post-fire. No plots sampled were in the early development stage pre-fire, although 23 plots transitioned into this development stage due to fire effects. The pre-fire landscape was primarily composed of mid-development stage patches (of both open and closed canopy stand structure) at 71 percent of all sampled plots, while late development stage patches of both canopy closures comprised the remaining plots. Post-fire, the landscape remained primarily in mid-development stage patches, though the relative proportion of this development stage was reduced to 55 percent of sampled plots. Likewise, post-fire late development stage patches were reduced in relative abundance to 26 percent of sampled plots. Closed canopy patches of any development stage (59%) were more abundant than open canopy patches pre-fire (41%), but open canopy patches (65%) were four times more abundant than closed canopy patches (16%) post-fire.

3.3 Predictors of Fire-induced Ecological Transitions

Classification tree analysis of all 27 potential predictor variables revealed that percent tree mortality was the greatest predictor of ecological transitions due to fire, followed by plot basal area (Figure 4). All 20 plots that had greater than 89 percent tree mortality transitioned to early development stage, whereas most plots (40 of 43) that experienced less than 31 percent tree mortality maintained the current successional state. For plots that experienced between 31 and 89 percent tree mortality, ecological transitions were best determined by the pre-fire basal area. Plots with pre-fire basal area less than 2.6 m^2 per plot were more likely to maintain the current ecological state (14 of 21 plots) and plots with a pre-fire basal area greater than 2.6 m^2 per plot were more likely to transition from closed to open canopy structure (30 of 38 plots). The final tree was pruned to three nodes and had an overall misclassification rate of 14.8 percent. The original tree had eight nodes and a misclassification rate of 9.8 percent (Appendix G).

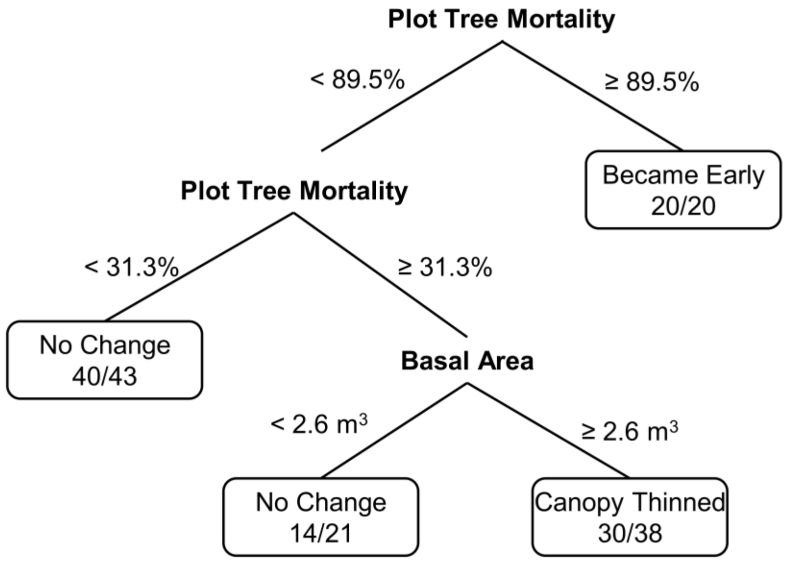

Figure 4: Classification tree for ecological transitions due to wildfire, based on 27 independent predictor variables of topography, vegetation, fire weather, and fire effects. Misclassification rate is 14.8%.

3.4 Predictors of Post-fire Development Stage

Classification tree analysis of post-fire development stage revealed that percent tree mortality was the greatest predictor variable, followed by average tree DBH and aspect (Figure 5). All plots that had greater than 89 percent tree mortality were early-development stage post-fire. For the remaining 102 plots, all 16 plots that had average DBH greater than 37 cm were late-development stage post-fire. Plots that had an average DBH between 34 and 37 cm were late-development stage if aspect was between 237 and 303 degrees (6 of 6 plots) and were mostly mid-development stage for all other aspects (7 of 9 plots). For plots that had an average DBH less than 34 cm, development stage post-fire was primarily mid-development (60 of 71 plots). The pruned tree had five nodes and a misclassification rate of 10.6 percent. The original tree had eight nodes and a misclassification rate of 5.7 percent (Appendix H).

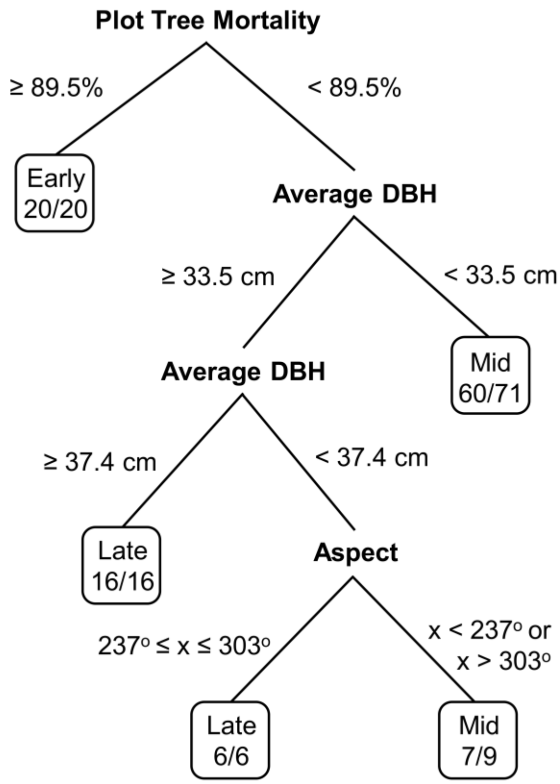

Figure 5: Classification tree for post-fire development stage, based on 27 independent predictor variables of topography, vegetation, fire weather, and fire effects. Misclassification rate is 10.6%.

4. Discussion

4.1 Major Findings

We found that the number of refugial plots in the study area was greatly reduced as a result of the 2012 wildfires when solely analyzing the degree of fire effects on the Camp et al. (1997) historic wildfire refugia. However, under our refined definition of wildfire refugia through examining fire-induced ecological transitions, the amount of plots that functioned as refugia in the study area (i.e. plot maintained successional state) was much higher (50%) than what Camp et al. had estimated occurred historically (20%). This result shows that even under extreme weather conditions and with historic refugial plots experiencing a higher degree of fire effects than non-refugial plots, there was no reduction in the study area of patches that functioned as refugia in the 2012 fires.

4.2 Comparison of Fire Effects between Sample Years

The number of unburned sampled plots post-fire was low compared to the proportion of plots Camp et al. classified as refugial, and classified refugial plots experienced a greater degree of fire effects than non-refugial plots particularly in the percentage of tree foliage scorched and the mortality of subcanopy trees. Although the Camp et al. refugial plots were not ‘minimally affected’ by fire in the 2012 fire events, this result is nonetheless consistent with Camp et al.’s concept of historic wildfire refugia experiencing a different fire regime than the surrounding matrix; historically, refugial patches burned with a lower frequency but higher severity fire regime in comparison to the higher frequency and lower severity fire regime of the non-refugial matrix. When historic refugial patches would eventually burn, as posited by Camp et al. (1997), refugial patches would burn at a higher severity than the surrounding matrix. Because of the longer fire return intervals in refugial patches, fire-sensitive and shade-tolerant tree species such as grand fir are able to establish and develop dense interconnected canopy layers with elevated fuel build-up (Hessburg et al. 2000), a concept which Camp et al. termed ‘outgrowing’ refugial status. The vegetative characteristics of these ‘outgrown’ refugia then led such patches to burn at a higher severity than the surrounding matrix.

4.3 Distribution of Post-fire Successional States and Ecological Transitions

The distribution of fire-induced ecological transitions displays the varying severity of fire effects that the study area experienced in 2012, which is consistent with forests in the inland northwest having historically been a mosaic of successional states and canopy structures that interchange with non-equilibrium dynamics (Agee 2005, Hessburg et al.

2007, Stine et al. *in press*). Ecological transitions due to fire reflect the different degrees of burn severity, where plots that persisted in the pre-fire successional state experienced low severity fire effects, plots with canopy thinning experienced mixed severity fire effects, and plots that transitioned to the early successional state experienced high severity fire effects (Keeley 2009). The proportion of observed 2012 fire-induced ecological transitions and associated burn severities match very closely with estimates of the proportion of historical burn severities compiled for the east Cascade forests of the inland northwest; we found that a respective 50%, 31%, and 19% of our sampled plots experienced low, mixed, and high severity fire effects respectively, compared to the historical estimate of 45%, 31%, and 24% of the landscape experiencing low, mixed, and high severity fire effects respectively (Hessburg et al. 2007). In general, fire was historically frequent in the Wenatchee National Forest (Everett et al 2000, Wright and Agee 2004), which created a patchy mosaic of mixed successional states and canopy structures due to the complex nature of the mixed severity fire regime (Agee 2005).

We found less than 30 percent of plots resampled in 2014 were late development stage pre-fire with mid-development stage plots being most common instead, which is consistent with Hessburg et al.'s (2007) findings of the historical landscape composition dominated by mid-development stage stands. We encountered no plots that were early successional pre-fire, which was unsurprising given that prior to 2012 the most recent wildfire in the study area occurred before 1900 (Holstine 1992). While persistence in development stage post-fire was high for both late and mid-development stage plots, the slightly higher rate of stand-replacing fire in mid-development stage plots shifted the landscape mosaic towards a more even distribution of development stages, which is closer to the historical landscape distribution even considering the relative dominance of mid-development stage patches (Hessburg et al. 2005, Hessburg et al. 2007). Post-fire 19 percent of sampled plots transitioned to the early successional state, which increased the structural heterogeneity of the study area (Swanson et al. 2011), whereas without periodic fire occurrence forest patches are able to reach mid- and late development stages and remain in these development stages in the absence of fire or other ecosystem processes (Hessburg et al. 2007). Open canopy stands were four times more abundant than closed canopy stands post-fire, whereas closed canopy stands were more abundant pre-fire. This fire-induced transition to increased open canopy structure is consistent with mixed severity fires reducing canopy closure (Barrett 1988, Wright and Agee 2004, Hessburg et al. 2005), and is also consistent with forest series that have historically had higher proportions of open canopy structure (Agee 1993).

4.4 Ecological Predictors of State Transitions and Post-fire Development stages

The most important factor that determined the distribution of fire-induced ecological transitions and post-fire development stages was total plot tree mortality. Tree mortality is a metric of burn severity resulting from fire behavior, which demonstrates that the cumulative effect of vegetation, topography, and fire weather acted synergistically to cause the observed ecological changes most relevant to post-fire states and transitions; thus, no individual component of the fire behavior triangle was the dominant overriding predictor. Our unpruned classification trees included all three categories of predictor variables from the fire behavior triangle (Appendices G & H), but since most of these variables occurred as lower splits on the classification tree and were mostly pruned out of the final models, these variables were more explanatory of local variations in fire effects for the study area and thus are not more broadly generalizable to a predictive model of fire effects for other areas. The primary importance of tree mortality in predicting post-fire states and transitions is not surprising because the vegetative attributes of the plots (maximum DBH, average DBH, canopy height, and canopy closure) were all variables used in classifying the pre- and post-fire successional states and subsequent fire-induced ecological transitions of the plots. High tree mortality due to fire directly influences these vegetative attributes and can cause a change in the successional state classification of the plot.

Aside from the tree mortality variable, the next strongest predictor variables in either classification tree were vegetative (average DBH and pre-fire plot basal area). Plots with larger trees were more likely to be late development stage post-fire; plots with a lower pre-fire basal area were less likely to change successional state post-fire while plots with a higher pre-fire basal area were more likely to transition from closed to open canopy stand structure. This is consistent with research in temperate forests showing that higher density stands with fewer large diameter trees experience more severe fire effects including greater tree mortality and canopy loss than lower density stands with a higher number of large diameter trees (Alexander et al. 2006, Collins and Stephens 2007, Miller et al. 2012, Lutz et al. 2013).

The only topographic variable included in either of our pruned classification trees occurred as a fourth-order split in the post-fire development stage classification tree, where the split partitioned late-development from mid-development plots based on a western aspect between 237 and 303 degrees. The specificity of this aspect criterion may potentially be an artifact of sampling, because the majority of plots sampled occurred on a western aspect. However, this result supports the evidence that topography contributes to refugia formation through modulating vegetative characteristics and fire behavior, but only as a subsidiary driver (Heyerdahl et al. 2001, Kushla and Ripple 2007). Similarly, Camp et al. (1997) found mixed results in terms of topographic location of historic wildfire refugia in the study area; the topographical parameters identified as the greatest predictor of refugial status still only contained 49 percent classified refugia on such sites, which further demonstrates the influences external to topography in the development of wildfire refugia.

If the extreme fire weather conditions seen during the 2012 fires overcame other contributing factors to burn severity as occurs on some wildfires (Bessie and Johnson 1995, Agee 1997), we would have expected weather variables to occupy higher splits of the classification tree. However, no fire weather variables were included in the final classification trees. Likewise, for their nearby North Cascades study area, Pritchard and Kennedy (2014) found that even during extreme weather, non-weather factors strongly influenced the pattern of burn severity. The effects of vegetation alteration due to fire suppression have also been found to be a greater influence on contemporary burn severity than climate or weather drivers (Hanson and Odion 2014). Thus, even under extreme weather conditions, fire weather factors alone are a poor predictor of post-fire successional states and transitions for this ecosystem. However, in mesic temperate environments like the inland northwest, fire cannot occur unless certain fire weather thresholds are reached (Agee 1997); this implies that fire weather variables have an implicit role in enabling a fire to burn, though local variability in fire effects is still more strongly driven by non-weather factors.

4.5 Did Fire Effects from the 2012 Wildfires Differ from Historical Normal?

Although fires may be becoming more frequent and larger across the landscape as a whole (Littell et al. 2009, Abatzoglou and Kolden 2013), and climate change is likely associated with more severe fire weather conditions (e.g., Westerling et al. 2006, Jones et al. 2009, Abatzoglou and Kolden 2011), the severity of the 2012 wildfires in our study area does not appear to significantly deviate from historical fire effects based on observed ecological transitions in the 2012 fires when compared to fire effects described in the literature. These contemporary fires induced varying levels of ecological change, which is consistent with the historical range of variability for the mixed-severity fire regime (Wright and Agee 2004, Agee 2005, Hessburg et al. 2007). In contrast to claims of increasing high-severity fire shown in the Sierra Nevada (Miller et al. 2012, Miller and Safford 2012), a growing body of research across different study areas has shown that contemporary fire effects either match historical proportions (Odion et al. 2004, Collins and Stephens 2007, Odion et al. 2014), or that the proportion of high severity burned area is not increasing (Miller et al. 2012, Hanson and Odion 2014). This opportunistic resample of the Camp et al. (1997) study plots corroborates the body of evidence for historically-representative burn severity for a novel forest ecosystem in the inland northwest.

4.6 Limitations of Study

We were unable to use the specific pre-fire vegetative information preserved by Camp et al. (1997) which necessitated using reconstructed data about stand structure pre-fire. This limited our ability to directly compare observed fire effects to Camp et al.'s original plot measurements in a statistically robust paired-plot format and also necessitated that we reconstruct pre-fire vegetation with only burned post-fire vegetation available to sample, which can be problematic due to uncertainties of estimating pre-fire vegetation with limited observable evidence (Morgan et al. 2014). Canopy cover is particularly difficult to measure from ocular estimates as well (Korhonen et al. 2006), especially after the canopy has been partially consumed in a fire. For this reason, we focused our research analysis on our post-fire successional state classifications rather than our reconstructed pre-fire successional states. Additionally, because we could not relocate the Camp et al. sampled trees, we were not able to use the ages of trees determined by Camp et al. in our analysis of fire effects. The age of a stand was an integral component of the Camp et al. refugial classification, and larger trees were more often indicative of more productive non-refugial sites rather than late development refugial sites (Camp, *personal communication*).

A number of our variables used in successional state classification and classification tree analysis were derived from only the sampled trees (n=10) in each plot. Though surveying the entire population of trees on a sample plot would have given the most accurate information about the vegetative structure and composition of each plot, we opted for the systematic 10 tree sub-sample in order to increase the geographic extent and number of plots we could visit given time and resource constraints. Additionally, we sampled only the portion of the entire 25,274 ha fire area which was originally sampled by Camp et al. (1997), which limits our results to the extent of the study area instead of the broader fire area.

4.7 Implications and Applications for Management

Many forest species are fire-sensitive and require refugial habitat to persist in the landscape (Agee 1993), including species of great management concern such as the northern spotted owl (*Strix occidentalis* caurina) which was a focal species of Camp et al.'s refugia concept. In order to alter fire behavior to reduce tree mortality and to promote heterogeneous fire effects (including refugia development and retention), mangers can apply common vegetation manipulation methods (Agee and Skinner 2005, Schwilk et al. 2009, Stephens et al. 2009). Because post-fire ecological transitions observed in this study were primarily associated with burn severity factors and forest density metrics (as opposed to factors that cannot feasibly be controlled such as topography or weather), our findings suggest that existing recommendations for altering fire behavior and severity will also serve to manage and control transitions between successional states. For example, fuel reduction treatments such as thinning (basal area reduction) and increasing height-to-live crown are frequently promoted to reduce fire effects such as crown fire initiation and fire residence time (Graham et al. 1999). The spatial heterogeneity of trees within a plot also affects the vulnerability of a patch during a fire event, which allows managers to create spatial patchiness and clumps of trees at a stand-level scale to further increase resilience to wildfire (Larson and Churchill 2012). However, as demonstrated by our study, even unmanaged forests that burn under extreme weather conditions can still have a high proportion of the landscape that remains refugial for a variety of different successional states. Managers can actively manage forest stand vegetation to either promote or diminish proportions of specific successional states in the landscape, but areas that function as refugia will tend to remain present in the landscape without management action as well.

In the face of environmental change, managers can take actions to increase the resilience of an ecosystem (Walker et al. 2004), and there have been calls for restoration of forest ecosystems throughout the western United States as well (Rasmussen et al. 2012). Because the observed fire effects of the 2012 fires were very similar to the historical range of variability observed for this ecosystem, actions that allow fire to naturally return to the ecosystem will likely aid in the two-fold management goal of increased resilience and forest restoration. As a caveat, Haugo et al. (2015) found that the east Cascades forests of the inland northwest had the lowest restoration need for all eco-regions studied, which implies that the universal return of fire to all fire-adapted ecosystems may not produce fire effects within the historical range of variability; although the return of fire holds great promise for restoration purposes, more studies on fire effects for other ecosystems are still needed. Regardless of the lower relative restoration needs in the east Cascades, restoration is still appropriate as our study and other studies (Hessburg et al. 2005, Haugo et al. 2015) have demonstrated lower than historical proportions of both early development and late development habitat, with higher proportions of both mid-development habitat and closed canopy structure patches across the landscape. Well managed fire in these ecosystems can be an effective management tool to thin the forest and reduce canopy closure, as well as creating early successional habitat (Haugo et al. 2015). Though late development habitat patches can only develop when given enough time, managers can take actions to encourage late development stage patches to develop and be retained in future fires in order to increase the proportion of late development habitat in the landscape.

Historic mixed-severity fire regimes can be used by managers as spatial and temporal models for the historical composition of the landscape for management and restoration goals (Everett et al. 2000, Agee 2005, Hessburg et al. 2007). By keeping wildfire in the landscape, spatial heterogeneity and patchiness will be created and maintained, which will reduce the vulnerability that future fires may burn with uncharacteristic severity patterns (Everett et al. 2000, Odion et al. 2014), especially in light of climate change. In contexts where wildfire cannot feasibly be used in management scenarios (e.g., wildland-urban interface), the use of 'fire surrogate' vegetation management which mimics fire-induced spatial heterogeneity should be employed (Agee 2005). Additionally, further research and quantification of fire effects following contemporary wildfires will also help continuously improve quantitative landscape composition models (such as VDDT) in addition to determining when forests have become vulnerable towards a permanent transition to a completely new vegetation state (Smith et al. 2014).

5. Conclusions

The formation of wildfire refugia and forest patches that are able to persist in the same successional state after a fire event is driven by the complex interaction of vegetation, topography, and fire weather that result in a spectrum of higher or lower levels of burn severity. Although historic wildfire refugia identified by Camp et al. did not remain refugial in the 2012 wildfires according to the higher degree of fire effects than the non-refugial matrix, much more of the study area as a whole could be considered refugial according to plots that persisted in the pre-fire successional state after the fire event. Persistent patches occurred even though these wildfires burned under extreme fire weather conditions, and the majority of the sampled area experienced low or mixed severity fire with only a small proportion experiencing high severity stand-replacement fire. Such heterogeneity of fire effects in this ecosystem resulted in a landscape mosaic of successional states that is consistent with published estimates of the historical landscape mosaic and the historical range of variability of successional states and ecological transitions for this ecosystem. Ultimately, this research demonstrates that there may not be such a thing as *permanent* wildfire refugia, but that areas which function as refugia can be maintained in the landscape as a whole during fire events. These non-permanent wildfire refugia may rotate spatially around the landscape mosaic based on the successional development of vegetation and the different types of fire regimes and return intervals that maintain impermanent wildfire refugia.

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6.2 Data Sources

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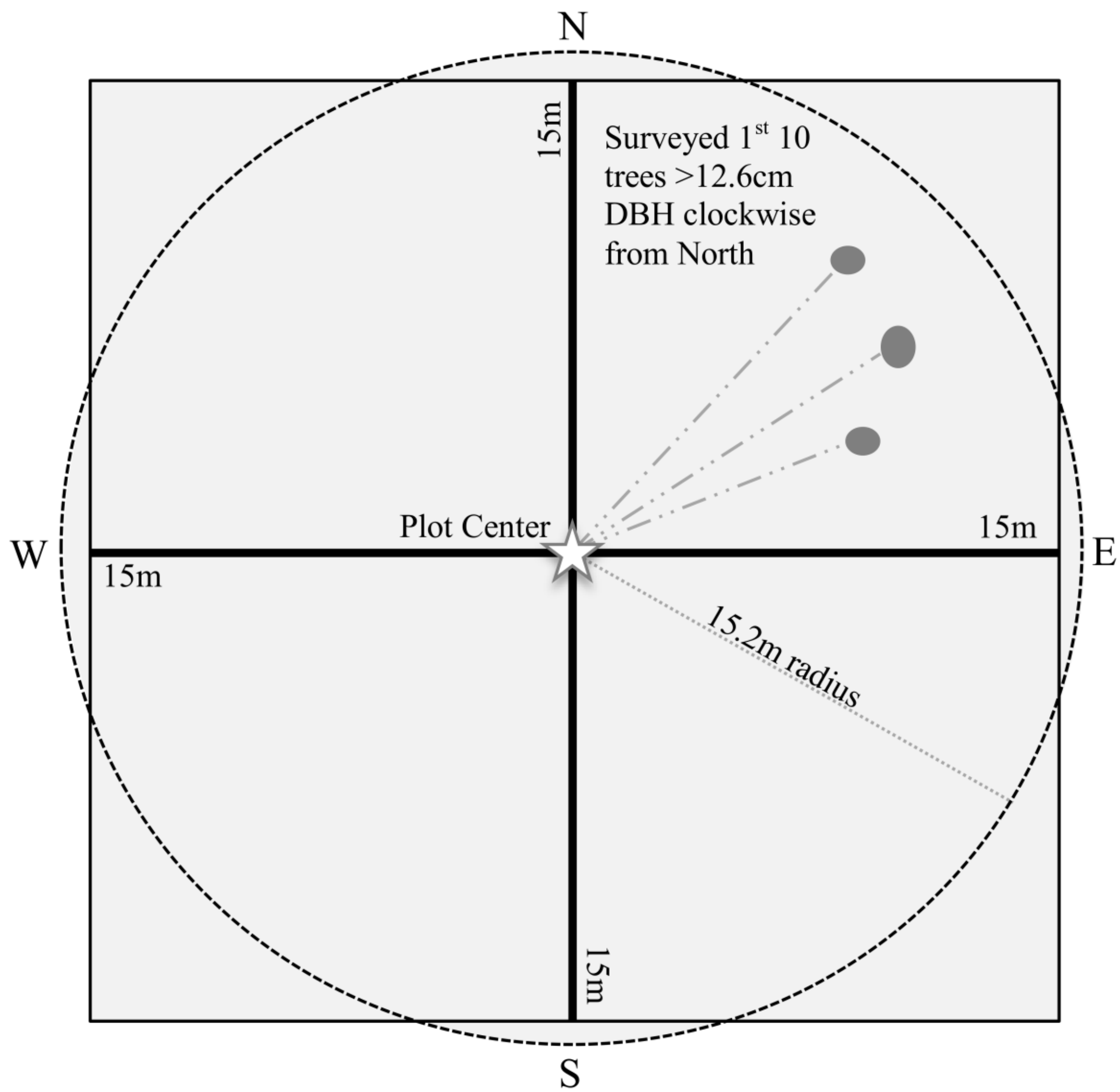
Appendix A: Distribution of refugial and non-refugial plots by drainage

Drainage	Refugial	Non-Refugial	All Plots
Burned in 2012	43	183	226
Resampled in 2014	41	81	122
Boulder Drainage Burned	22	94	116
Boulder Drainage Resampled	22	48	70
Mission Drainage Burned	6	37	43
Mission Drainage Resampled	4	12	16
Tronsen Drainage Burned	15	52	67
Tronsen Drainage Resampled	15	21	36

Appendix B: Diagram of 2014 field sampling protocol.



Appendix C: Camp et al. (1997) classifications for topography type variable.

Code	Topography Type
A	Stream Confluence
B	On Ridge
C	Lower 1/3 of Slope
D	Middle 1/3 of Slope
E	Upper 1/3 of Slope
F	Bench
G	Lower Headwall
H	Middle Headwall
I	Upper Headwall
J	In Channel

Appendix D: Distribution of VDDT assigned successional classes according to BpS model, VDDT estimated representative landscape proportion, and departure from VDDT proportion.

	Pre-fire State			Post-fire State			VDDT Proportion
	Count	Percent	Departure	Count	Percent	Departure	
Northern Rocky Mountain Dry-Mesic Montane Mixed Conifer Forest (n = 17)							
Early Development	0	0%	-10%	0	0%	-10%	10%
Mid-Development Closed Canopy	3	18%	+13%	3	18%	+13%	5%
Mid-Development Open Canopy	10	59%	+29%	10	59%	+29%	30%
Late Development Open Canopy	1	6%	-39%	2	12%	-33%	45%
Late Development Closed Canopy	3	18%	+8%	2	12%	+2%	10%
East Cascades Mesic Montane Mixed-Conifer Forest and Woodland (n = 92)							
Early Development	0	0%	-10%	13	14%	+4%	10%
Mid-Development Closed Canopy	36	39%	+19%	9	10%	-10%	20%
Mid-Development Open Canopy	25	27%	+22%	42	46%	+41%	5%
Late Development Open Canopy	9	10%	-5%	22	24%	+9%	15%
Late Development Closed Canopy	22	24%	-26%	6	6%	-44%	50%
Rocky Mountain Subalpine Dry-Mesic Spruce-Fir Forest and Woodland (n = 13)							
Early Development	0	0%	-5%	10	77%	+72%	5%
Mid-Development Closed Canopy	8	61%	+41%	0	0%	-20%	20%
Mid-Development Open Canopy	4	31%	-9%	3	23%	-17%	40%
Late Development Open Canopy	1	8%	-17%	0	0%	-25%	25%
Late Development Closed Canopy	0	0%	-10%	0	0%	-10%	10%

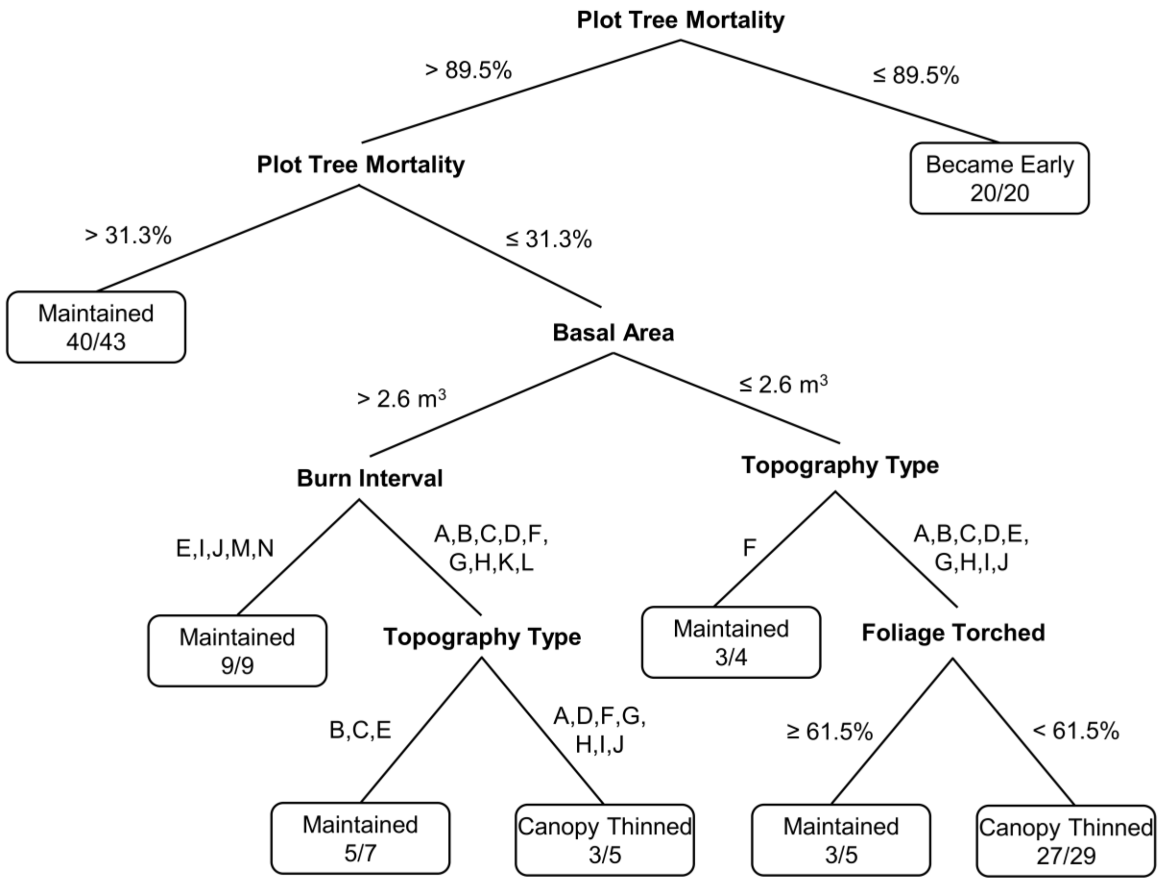
Appendix E: Table of 39 potential predictor variables for use in classification tree analysis

Category	Type	Name	Included in Classification Tree
Topographic	Quantitative	Aspect Eastness	Yes
		Aspect Northness	Yes
		Slope	Yes
		Elevation	Yes
	Qualitative	Topography Type	Yes
		Drainage	Yes
Vegetative	Quantitative	Maximum Canopy Height	Yes
		Average DBH	Yes
		Maximum DBH	No
		Pre-fire Plot Basal Area	Yes
		All Trees Pre-fire	Yes
		Overstory Trees Pre-fire	Yes
		Subcanopy Trees Pre-fire	No
		Total Canopy Cover Pre-fire	No
		Overstory Canopy Cover Pre-fire	No
		Subcanopy Canopy Cover Pre-fire	No
	Qualitative	Canopy Structure	Yes
		Abies grandis present	Yes
		Abies lasiocarpa present	Yes
		Larix occidentalis present	Yes
		Picea engellmannii present	Yes
		Pinus contorta present	Yes
		Pinus ponderosa present	Yes
		Populus spp. present	Yes
		Psuedotsuga menziesii present	Yes
		Mistletoe present	Yes
Fire Weather	Quantitative	Average Wind Speed	No
		Maximum Wind Speed	Yes
		Maximum Temperature	No
		Minimum Relative Humidity	Yes
	Qualitative	Burn Interval	Yes
Fire Effects	Quantitative	Percent Bole Char	No
		Average Char Height	Yes
		Average Foliage Scorch	No
		Average Foliage Torch	Yes
		Total Tree Mortality	Yes
		Overstory Tree Mortality	No
		Subcanopy Tree Mortality	No
		Total Plot CBI Score	No

Appendix F: Covariance matrix of 26 potential quantitative predictor variables. Variables removed due to high covariance denoted with asterisk.

	Aspect Eastness	Aspect Northness	Slope	Elevation	Canopy Height	Average DBH	Maximum DBH*	Basal Area	All Trees Prefire	Overstory Trees Prefire	Subcanopy Trees Prefire*	Total Canopy Cover*	Overstory Canopy Cover*	Understory Canopy Cover*	Average Wind*	Maximum Wind	Maximum Temp*	Minimum RH	Percent Bole Char*	Average Char Height	Foliage Scorch*	Foliage Torch	Total Tree Mortality	Overstory Tree Mortality*	Subcanopy Tree Mortality*	CBI*
Aspect Eastness	1.000	0.030	0.265	-0.077	-0.270	-0.399	-0.244	-0.103	0.248	0.014	0.293	-0.063	-0.196	0.095	-0.241	-0.275	0.201	-0.115	0.101	0.181	0.302	0.060	0.309	0.204	0.283	0.222
Aspect Northness	0.030	1.000	0.360	-0.160	-0.071	-0.063	-0.023	-0.252	-0.184	-0.159	-0.173	-0.227	-0.217	-0.182	-0.367	-0.401	-0.033	-0.096	-0.155	-0.122	-0.172	0.025	-0.292	-0.150	-0.341	-0.218
Slope	0.265	0.360	1.000	0.201	-0.244	0.032	-0.027	-0.226	-0.192	-0.214	-0.188	-0.289	-0.256	-0.234	-0.311	-0.274	0.327	-0.314	0.135	0.229	-0.010	0.140	0.032	0.084	0.015	0.103
Elevation	-0.077	-0.160	0.201	1.000	0.070	0.349	0.298	0.182	-0.096	0.035	-0.157	-0.038	0.108	-0.182	-0.151	-0.125	0.521	-0.220	0.011	0.205	0.048	0.188	0.066	0.160	0.057	0.163
Canopy Height	-0.270	-0.071	-0.244	0.070	1.000	0.347	0.356	0.325	-0.020	-0.047	0.062	0.167	0.204	0.114	0.223	0.021	-0.065	0.046	0.187	0.108	0.027	0.040	0.039	-0.075	0.085	0.118
Average DBH	-0.399	-0.063	0.032	0.349	0.347	1.000	0.842	0.374	-0.483	-0.183	-0.539	-0.153	0.130	-0.410	-0.004	0.018	0.244	-0.222	0.161	0.203	-0.136	0.118	-0.128	-0.060	-0.050	0.020
Maximum DBH*	-0.244	-0.023	-0.027	0.298	0.356	0.842	1.000	0.298	-0.437	-0.274	-0.424	-0.210	-0.019	-0.359	0.014	0.037	0.308	-0.264	0.080	0.192	-0.101	0.250	-0.097	-0.040	-0.057	0.017
Basal Area	-0.103	-0.252	-0.226	0.182	0.325	0.374	0.298	1.000	0.581	0.616	0.440	0.675	0.680	0.402	-0.063	-0.052	0.105	0.018	0.290	0.206	0.150	-0.108	0.334	0.287	0.410	0.413
All Trees Prefire	0.248	-0.184	-0.192	-0.096	-0.020	-0.483	-0.437	0.581	1.000	0.727	0.907	0.761	0.500	0.753	-0.105	-0.094	-0.082	0.186	0.063	-0.025	0.182	-0.241	0.382	0.243	0.391	0.319
Overstory Trees Prefire	0.014	-0.159	-0.214	0.035	-0.047	-0.183	-0.274	0.616	0.727	1.000	0.418	0.748	0.823	0.369	-0.191	-0.126	-0.068	0.155	0.072	-0.037	0.099	-0.320	0.193	0.192	0.286	0.222
Subcanopy Trees Prefire*	0.293	-0.173	-0.188	-0.157	0.062	-0.539	-0.424	0.440	0.907	0.418	1.000	0.612	0.229	0.835	0.025	-0.004	-0.097	0.190	0.082	0.004	0.220	-0.123	0.446	0.239	0.401	0.331
Total Canopy Cover*	-0.063	-0.227	-0.289	-0.038	0.167	-0.153	-0.210	0.675	0.761	0.748	0.612	1.000	0.827	0.767	-0.077	-0.084	-0.071	0.176	0.077	-0.079	0.075	-0.285	0.237	0.169	0.290	0.245
Overstory Canopy Cover*	-0.196	-0.217	-0.256	0.108	0.204	0.130	-0.019	0.680	0.500	0.823	0.229	0.827	1.000	0.332	-0.120	-0.080	-0.029	0.136	0.064	-0.068	-0.007	-0.277	0.072	0.093	0.185	0.155
Subcanopy Canopy Cover*	0.095	-0.182	-0.234	-0.182	0.114	-0.410	-0.359	0.402	0.753	0.369	0.835	0.767	0.332	1.000	0.091	0.007	-0.162	0.234	0.099	-0.053	0.146	-0.142	0.355	0.210	0.312	0.267
Average Wind*	-0.241	-0.367	-0.311	-0.151	0.223	-0.004	0.014	-0.063	-0.105	-0.191	0.025	-0.077	-0.120	0.091	1.000	0.851	-0.367	0.220	0.112	0.034	0.073	0.213	0.073	-0.038	0.063	-0.008
Maximum Wind	-0.275	-0.401	-0.274	-0.125	0.021	0.018	0.037	-0.052	-0.094	-0.126	-0.004	-0.084	-0.080	0.007	0.851	1.000	-0.145	0.019	0.115	0.042	0.094	0.290	0.132	0.009	0.153	0.073
Maximum Temp*	0.201	-0.033	0.327	0.521	-0.065	0.244	0.308	0.105	-0.082	-0.068	-0.097	-0.071	-0.029	-0.162	-0.367	-0.145	1.000	-0.796	0.115	0.290	0.182	0.248	0.248	0.278	0.236	0.315
Minimum RH	-0.115	-0.096	-0.314	-0.220	0.046	-0.222	-0.264	0.018	0.186	0.155	0.190	0.176	0.136	0.234	0.220	0.019	-0.796	1.000	-0.109	-0.200	-0.109	-0.253	-0.142	-0.116	-0.155	-0.196
Percent Bole Char*	0.101	-0.155	0.135	0.011	0.187	0.161	0.080	0.290	0.063	0.072	0.082	0.077	0.064	0.099	0.112	0.115	0.115	-0.109	1.000	0.791	0.639	0.322	0.660	0.514	0.715	0.737
Average Char Height	0.181	-0.122	0.229	0.205	0.108	0.203	0.192	0.206	-0.025	-0.037	0.004	-0.079	-0.068	-0.053	0.034	0.042	0.290	-0.200	0.791	1.000	0.712	0.547	0.681	0.619	0.661	0.794
Foliage Scorch*	0.302	-0.172	-0.010	0.048	0.027	-0.136	-0.101	0.150	0.182	0.099	0.220	0.075	-0.007	0.146	0.073	0.094	0.182	-0.109	0.639	0.712	1.000	0.344	0.813	0.663	0.753	0.786
Foliage Torch	0.060	0.025	0.140	0.188	0.040	0.118	0.250	-0.108	-0.241	-0.320	-0.123	-0.285	-0.277	-0.142	0.213	0.290	0.248	-0.253	0.322	0.547	0.344	1.000	0.322	0.296	0.274	0.386
Total Tree Mortality	0.309	-0.292	0.032	0.066	0.039	-0.128	-0.097	0.334	0.382	0.193	0.446	0.237	0.072	0.355	0.073	0.132	0.248	-0.142	0.660	0.681	0.813	0.322	1.000	0.786	0.947	0.907
Overstory Tree Mortality*	0.204	-0.150	0.084	0.160	-0.075	-0.060	-0.040	0.287	0.243	0.192	0.239	0.169	0.093	0.210	-0.038	0.009	0.278	-0.116	0.514	0.619	0.663	0.296	0.786	1.000	0.691	0.776
Subcanopy Tree Mortality*	0.283	-0.341	0.015	0.057	0.085	-0.050	-0.057	0.410	0.391	0.286	0.401	0.290	0.185	0.312	0.063	0.153	0.236	-0.155	0.715	0.661	0.753	0.274	0.947	0.691	1.000	0.896
CBI*	0.222	-0.218	0.103	0.163	0.118	0.020	0.017	0.413	0.319	0.222	0.331	0.245	0.155	0.267	-0.008	0.073	0.315	-0.196	0.737	0.794	0.786	0.386	0.907	0.776	0.896	1.000

Appendix G: Unpruned classification tree for fire-induced ecological transitions



Appendix H: Unpruned classification tree for post-fire development stage