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Fire Effects on Historical Wildfire Refugia in Contemporary Wildfires

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Abstract: Wildfire refugia are forest patches that are minimally-impacted by fire and provide critical habitats for fire-sensitive species and seed sources for post-fire forest regeneration. Wildfire refugia are relatively understudied, particularly concerning the impacts of subsequent fires on existing refugia. We opportunistically re-visited 122 sites classified in 1994 for a prior fire refugia study, which were burned by two wildfires in 2012 in the Cascade mountains of central Washington, USA. We evaluated the fire effects for historically persistent fire refugia and compared them to the surrounding non-refugial forest matrix. Of 122 total refugial (43 plots) and non-refugial (79 plots) sites sampled following the 2012 wildfires, one refugial and five non-refugial plots did not burn in 2012. Refugial sites burned more severely and experienced higher tree mortality than non-refugial plots, potentially due to the greater amount of time since the last fire, producing higher fuel accumulation. Although most sites maintained the pre-fire development stage, 19 percent of sites transitioned to Early development and 31 percent of sites converted from Closed to Open canopy. These structural transitions may contribute to forest restoration in fire-adapted forests where fire has been excluded for over a century, but this requires further analysis.

Keywords: burn severity; forest structure; succession; Cascade Range; restoration; mixed-conifer forest

1. Introduction

Wildfire is an integral natural ecosystem process worldwide [1], including the conifer forests of the inland of northwestern United States [2]. In these fire-adapted ecosystems, not every stand is affected equally by wildfire [3–5] and differential degrees of ecological change can result from variation in burn severity within single fires [6]. Spatially heterogeneous burn severity drives forest structure and composition [7,8] and produces complexity that is critical to supporting biodiversity, ecosystem services, and forest resilience [2,9,10]. Much of the inland Northwest is characterized by a mixed-severity fire regime that includes both frequent, low-severity fire and infrequent, high-severity fire [2]. Less studied, but equally important ecologically, are forest patches that remain unburned or experience a relatively low degree of change from a wildfire event; these patches are often described as 'wildfire refugia' [11–14]. Such patches represent key landscape elements that support the persistence of fire-sensitive flora and fauna both during the fire (as a refuge) and after the fire (as intact habitat), and provide seed sources for the regeneration of adjacent severely burned areas post-fire [14]. One of the key uncertainties about refugia is their persistence, and what characteristics determine persistent versus temporary refugia during successive ecological disturbances [11]. Resolving this uncertainty is necessary for several reasons, including improving our understanding and the predictability of forest succession dynamics, developing conservation management strategies for critical refugia that are vulnerable, and establishing a baseline from which global change impacts can be measured [14].

Refugia have been broadly defined and delineated, depending on the discipline and research query [15]; to-date, there is no single, widely-accepted definition of fire refugia [4,12,14]. Areas within fire perimeters that do not experience any fire effects, e.g., [16,17], or that experience fire effects at a lower severity than surrounding areas, e.g., [11], have both been described as wildfire refugia. In the literature, the scale and characteristics of wildfire refugia vary by organism and ecosystem [14], leading to different definitions of what constitutes a wildfire refugium. The occurrence of wildfire refugia depends upon several environmental factors that vary spatially and temporally, including topography, climate, soils, geomorphology, and ecological disturbances such as meteorological events, insects, pathogens, and fire [18,19]. These factors interact together to create spatial heterogeneity in vegetation and forest structure, which is maintained by fire [20]. For example, vegetation characteristics such as stand age or structure can either increase or decrease the likelihood of fire occurrence [21,22] or can minimize or exacerbate the effects when a fire does occur [23]. Additionally, topographic complexity influences burn severity [23,24] and refugia formation [11,12]. 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 [24–27], but in extreme fire weather events, local weather conditions may override vegetative and landscape effects on burn severity [28,29]. Furthermore, human activities such as stand management have the potential to influence the severity of burn patterns and the formation and persistence of refugial patches [30].

Changes in land cover and land use have greatly altered both fire frequency and intensity in inland northwest forests [31], further confounding our ability to understand the formation of fire refugia. Euro-American settlements and associated timber harvesting and grazing, along with a century of fire exclusion, have impacted the forest stand structure [32,33], as well as the spatial distribution and intensity of wildfires, increasing the risk of stand-replacing fire [5,25]. Inland northwest forests are already experiencing climatic conditions conducive to an increased occurrence and duration of wildfires (e.g., extended heat waves and droughts), and these trends are projected to continue through the 21st century [34–37]. Recent studies suggest that the severity of wildfires may be increasing for some ecosystems [38,39], although the robustness of such trends is questioned by other studies [27,40–42]. If the overall fire severity increases, fire refugia that were historically sheltered from fire or experienced only low severity fires might now burn at a higher severity under the contemporary fire regime. To-date, however, there has been little opportunity to assess the effects of the contemporary, altered fire regime on historically persistent fire refugia.

In 2012, the Wenatchee Complex Fires burned through an area of the central Washington state, USA, that had been previously sampled and classified into refugial and non-refugial patches two decades prior to the two fires [11]. This provided a unique opportunity to assess contemporary fire effects on field-delineated refugial and non-refugial stands to examine the persistence of historic wildfire refugia under changing fire regimes and climatic conditions. To our knowledge, no field-based studies have specifically assessed the persistence of wildfire refugia based on the pre-fire identification of long-term wildfire refugia. Camp and colleagues [11] classified historic wildfire refugia using forest stand structure, tree age, and species composition data within the Wenatchee National Forest, defining historic wildfire refugia as forest patches that had been minimally affected by fire events for at least 140 years (well before the onset of fire exclusion or active fire suppression [43]), while the surrounding forest matrix had experienced greater fire effects in historic fire events. The primary research objective of this study was to determine the effects of the 2012 wildfires on these historical fire refugia, specifically their persistence and changes in forest structure. This question was addressed by (1) quantifying and comparing the fire effects between refugial and non-refugial plots as classified pre-fire by Camp et al. [11] and (2) by investigating changes from inferred pre- to post-fire forest stand structure across stands that had been previously classified as refugial or non-refugial. More broadly, this case study evaluates how the occurrence of wildfire refugia and the mosaic of forest structure may shift in response to changing fire regime characteristics in the contemporary era. While conclusions from a case study are limited in application, such studies are needed to identify threats to critical refugia that are central to conservation plans for key species.

2. Materials and Methods

2.1. Study Area

The study area is located in the 47,794 ha Swauk Late Successional Reserve of the Okanogan-Wenatchee National Forest (Cle Elum Ranger District) of central Washington State, USA (Figure 1). A Late Successional Reserve (LSR) is a management designation for an area created through the Northwest Forest Plan with the objective of protecting and enhancing the condition of late-successional and old-growth forest ecosystems; as such, only limited stand management is permitted in LSR-designated areas [44]. Prior to attaining the 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 [11]. The study area is located at the far eastern edge of the Cascade mountain range, extending into the dry interior Columbia River Plateau to the east. Sampled plots in this study ranged in elevation from 1027 to 1912 m. Vegetative communities in the Swauk LSR form a heterogeneous landscape due to strong responses to the dissected topography, precipitation gradient, and insolation differences [45]. At more xeric sites, lower elevations, and south-facing aspects, open-canopy ponderosa pine (*Pinus ponderosa* Dougl. Ex Forbes) and Douglas-fir (*Psuedotsuga menziesii var. menziesii* (Mirb.) Franco) stands are common. At more mesic sites, higher elevations, and north-facing slopes, stands of subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.) and lodgepole pine (*Pinus contorta* var. *latifolia* Dougl. ex Loud.) are more typical. However, specific site conditions can cause the immediate juxtaposition of disjunct forest stands. Within the surveyed plots, grand fir (*Abies grandis* (Dougl. Ex D. Don.)) accounted for 39% of sampled trees, Douglas-fir accounted for 31%, and subalpine fir a further 11%.

The pre-European settlement fire regime near the Swauk LSR varied spatially by vegetative type with a mean fire return interval of seven to 43 years, and with large fires occurring approximately every 27 years [43]. Pre-European settlement fire severity also varied in the study area, with drier forest types experiencing low-severity fire, while more mesic forest types experienced occasional moderate and high-severity fires. However, fire regimes have been significantly altered since European settlement. Fire frequency declined dramatically around 1900, coinciding with the start of commercial logging [43] and the advent of active fire suppression in the Wenatchee National Forest [46]. Consequently, there is no evidence of fire occurrence in the Swauk LSR from 1900 to 2012 [47].

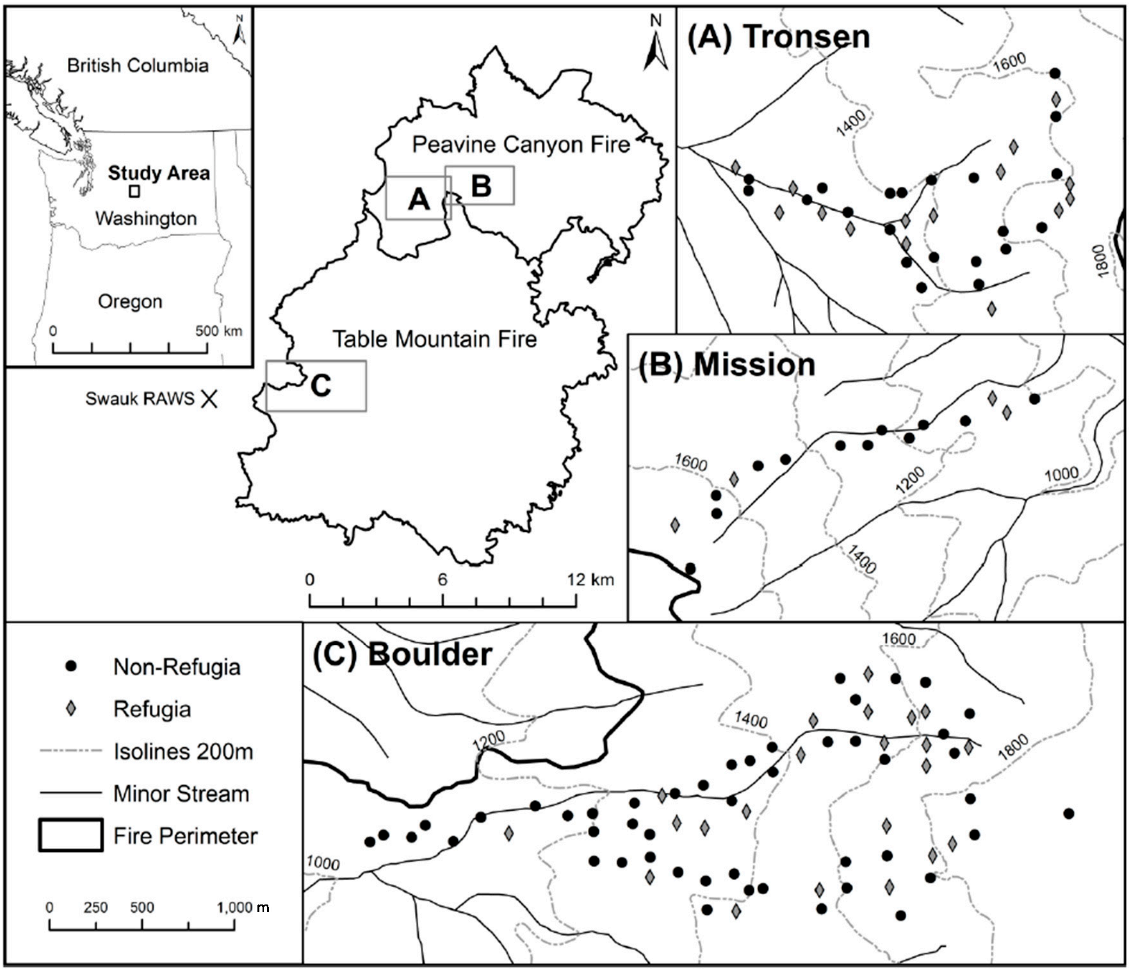

Figure 1. Study area in the east Cascades of central Washington State, USA, with burn perimeters of the 2012 Peavine Canyon and Table Mountain fires and spatial locations of the Camp et al. [11] classified refugial (diamonds) and non-refugial (circles) plots in the three study drainages (Tronsen, Mission, and Boulder).

2.2. The 2012 Fires

The Table Mountain and Peavine Canyon (as part of the Wenatchee Complex) fires burned simultaneously after the fires were ignited from lightning strikes in early September 2012. These fires eventually merged, creating a total burn area of 25,274 ha. These fires burned 226 sample sites of the Camp et al. [11] study in three different drainages, where 43 and 183 plots, respectively, were previously classified as refugial and non-refugial. 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 km west of the fire perimeter [48]. The average air temperature for the July through September fire season was higher than normal (83rd percentile), and the average air temperature for September when the fires ignited was much higher than normal (93rd percentile). The average relative humidity for the fire season was lower than normal (35th percentile), and the average relative humidity for September was much lower than normal (3rd 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 [48].

2.3. Field Measurements

Field data were collected in the summer and fall of 2014 by resampling the general locations of the Camp et al. [11] sample sites that burned in 2012 (Figure 1). The original Camp et al. [11] plots were established along multiple transects using a hip chain and sighting compass in the field, and the final plot locations were annotated on topographic maps. To determine the global positioning system (GPS) coordinates of these plots, the annotated topographic maps were first digitized into a geographic information system database and then plot coordinates were navigated to with a handheld GPS unit in the field. Since the prior plots were not permanently monumented, new plots were established as close as possible to the original plots within the same forest stand. GPS locations that fell in barren areas or in unsafe field sites were relocated to the nearest suitable forested site. A subset of plots from the Camp et al. [11] study was sampled across the entire elevational gradient for each drainage. A total of 41 refugial sites and 81 non-refugial sites were sampled across the three study drainages for a total of 122 sampled plots.

At each sample site, a modified replication of the Camp et al. [11] sampling protocol was conducted (Figure 2). Plot centers were monumented at the ascribed GPS coordinates and a 15.2-m diameter circular plot covering 725.8 m^2 was established. Variables collected or derived for the plot level at each site included four topographic, nine vegetative, and four fire effect variables (Table 1). Plot aspect was measured in degrees and then transformed into northness and eastness indices for analysis. We also collected variables at the tree level for each plot, where a sweeping transect from an azimuth of 0° (north) was used to sample the first ten trees in the plot meeting a minimum threshold diameter at breast height (DBH) of 12.6 cm. We tagged and assessed each set of 10 trees for three demographic and five fire effects variables (Table 2). 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. A total of 1220 trees were sampled in this study.

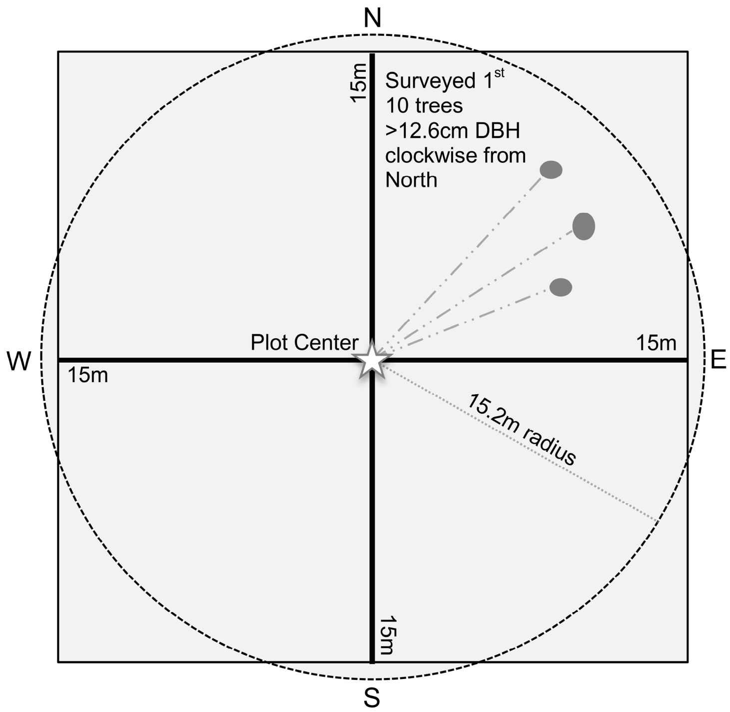

Figure 2. Diagram of the 2014 field sampling protocol.

Table 1. Plot-level topographic, vegetative, and fire effects variables.

Data Type	Variable	Definition
Topographic	Aspect Northness Aspect Eastness Slope Elevation Topography Type	cos(π/2-aspect); range from −1 (south) to 1 (north) sin(π/2-aspect); range from −1 (west) to 1 (east) Degrees; measured by clinometer Meters; measured by global positioning system Ten option categorical classification (from [11])
Vegetative	Max Canopy Height Species Present Canopy Structure All Trees Pre-fire	Meters; measured by Impulse Laser Dominant tree species Presence/absence of overstory/subcanopy strata Count of all trees alive in plot pre-fire
	Overstory Trees Pre-fire	Count of all ≥12.6 cm DBH 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 [π(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 ≥12.6 cm DBH tree mortality in overstory strata post-fire
	Subcanopy Tree Mortality	Count of all ≥12.6 cm DBH subcanopy tree mortality in subcanopy strata post-fire
	Total Plot CBI	Composite Burn Index protocol (score from 0 to 3)

DBH—diameter at breast height; CBI—Composite Burn Index [49].

Table 2. Tree-level demographic and fire effect variables.

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

Burn severity was also assessed at each site using the Composite Burn Index (CBI) protocol [49]. Due to the aggregation of fire effects in the CBI protocol [50], we also assessed the burn severity for each of the five different specific fire effect metrics at the tree level. Per [49], the CBI analysis area was modified to be a 30 by 30-m 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. [11] plot azimuths. While CBI is normally conducted one-year post-fire [49], the methodology has been previously utilized to assess fire effects two years post-fire [51]; doing so allows an assessment of longer-term fire effects while also capturing delayed mortality in the tree strata.

2.4. Data Quality Assurance

To ensure that the plots sampled in 2014 were comparable to the plots that Camp et al. [11] originally sampled and classified into potential refugia, we conducted a paired-plot assessment using the original plot data and stand delineations produced by Camp et al. [11]. Stand delineations developed by Camp et al. [11] and aerial imagery were used to determine if the original and resampled plots fell in the same stand. If the original and resampled plot pair was not visually within the same stand, then the topographic and vegetative attributes of each plot were compared for similarities. We were not able to confidently match 13 sampled plots through this qualitative comparison and these were excluded from further paired-plot analysis which resulted in a total of 109 confident paired-plot matches. To confirm that the plots were equivalent to Camp et al. [11], 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 [52] by calculating the modified Heat Load Index from McCune and Keon [53], using Equation (1) due to the steep slopes at our study site. As we found no significant differences, we contend that these 109 matched plots provide a conservative match to the Camp et al. [11] plot-level data.

2.5. Comparison of Fire Effects between Resampled and Original Plots

We used the 109 plots determined to be confident matches from the data quality assurance step (Section 2.4) and tested for differences in burn occurrence and burn severity between refugial (n = 36) and non-refugial (n = 73) plots as classified by Camp et al. [11]. Differences in burn occurrence between refugial and non-refugial plots were assessed with a chi-square test of independence.

Fire effects were compared between those plots which Camp et al. [11] classified as refugial versus non-refugial for eight fire effects metrics. Overall plot burn severity was assessed using the Composite Burn Index score and seven individual burn severity metrics (maximum bole char height, percent bole charred, percent foliage scorched, percent foliage torched and percent tree mortality of overstory, and sub-canopy tree strata) were compared between the Camp et al. [11] refugial and non-refugial plots using a Wilcoxon-signed-rank test due to the non-normal distribution of the data (α = 0.1). The Wilcoxon-signed-rank test results in a W-value where lower W-values correspond with lower p-values.

2.6. Assessment of Changes in Forest Structure

To investigate fire impacts on forest structure, we assigned successional states to each plot based on the inferred pre-fire and observed post-fire structural characteristics. In order to assign successional forest structure 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 [54,55]. BpS and other LANDFIRE vegetation products are widely used in state-and-transition studies because of the ecologically-based successional states and ecological transitions described in each BpS model [56–59]. All 122 plots sampled in 2014 were assigned to one of the three most common Biophysical Settings for the study area through an analysis of the 2014 quantitative plot data, recorded qualitative field observations, and photos for each plot. Since each BpS model 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 [60]; forest types were cross-walked to the Wenatchee National Forest correlate names of the Douglas-fir (n = 17), grand fir (n = 92), and subalpine fir series (n = 13) (Table 3), to be consistent with and comparable to the three forest series reported by Camp et al. [11] in their original analysis.

Table 3. Cross-walk between the BpS Model and Wenatchee NF Correlate from [60].

Biophysical Setting Model Name	Wenatchee NF Correlate	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 and forest series were assigned to each plot, pre- and post-fire forest structure was classified using the Vegetation Dynamics Development Tool (VDDT) [61] classes for each respective BpS model. The VDDT model for each of the three BpS models uses five distinct successional/structural classes (hereafter, referred to as "successional states"): Early Development, Mid-Development Open Canopy, Mid-Development Closed Canopy, Late Development Open Canopy, and Late Development Closed Canopy (Figure 3). Within each BpS model there are different attribute criteria for what constitutes a particular successional state based on species composition; quantitative criteria used in classification included canopy cover, canopy height, maximum tree size class, and average tree size class, while qualitative criteria included the species present, tree relative canopy position, and fuel model (Table 4). Post-fire successional state was classified through post-fire vegetation observed during the 2014 field season. For the pre-fire successional state, in-field estimates of canopy cover and counts of trees presumed living pre-fire were used as best approximations to infer pre-fire vegetative structure and composition. Field notes and plot photos were used to refine this successional state classification when quantitative data alone proved inconclusive.

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 the current successional state; (2) plot canopy was thinned from a closed to open canopy structure of the same development stage; or (3) plot transitioned to an early development successional state. We conducted the transition analysis for all 122 plots (not just the 109 paired plots) as we were not statistically comparing the refugial and non-refugial plots for this analysis, but rather characterizing changes in forest structure.

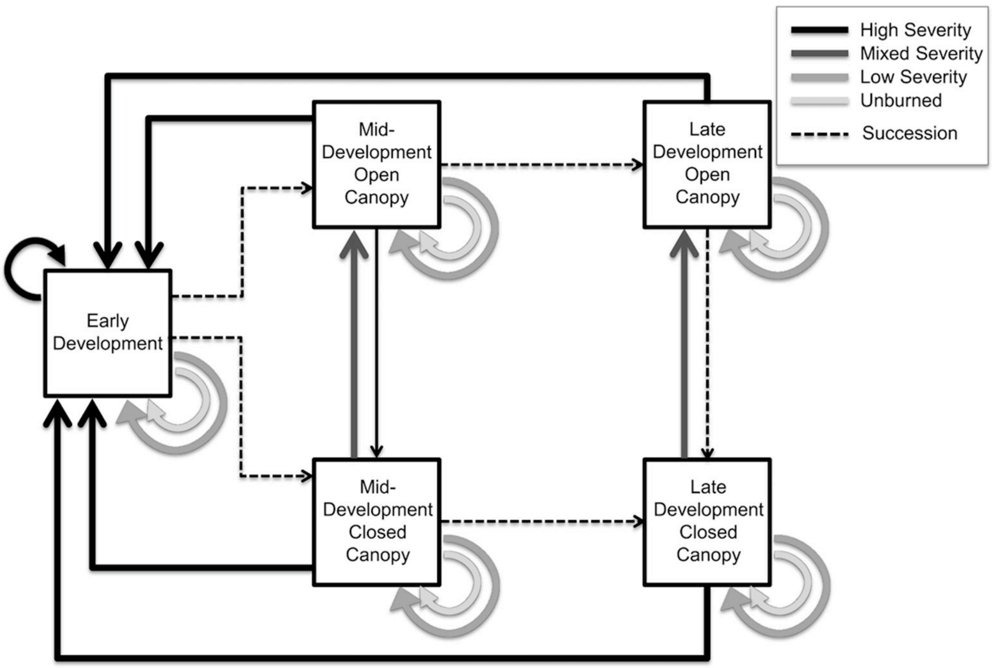

Figure 3. State-and-transition model of the Vegetation Dynamics Development Tool successional states and ecological transitions as affected by differential wildfire severity (modified from [4]).

Table 4. Quantitative criteria and thresholds utilized to classify plots into the five successional states that describe the forest structure for the three Biophysical Setting (BpS) models present in the study area. Both the initial classification of the plots into BpS models and further classification into successional states also utilized qualitative information and species lists in the BpS model descriptions, which can be found on the LANDFIRE website (www.landfire.gov).

Biophysical Setting (BpS)	State	Cover	Height	Tree Size Class
Northern Rocky Mountain Dry-Mesic Montane Mixed Conifer Forest (Douglas Fir Series)	Early	0–20%	Tree 0–5 m	Sapling > 4.5′, <5″ DBH
	Mid-Open	0–40%	Tree 5.1–25 m	9–21″ DBH
	Mid-Closed	41–100%	Tree 5.1–25 m	9–21″ DBH
	Late-Open	11–40%	Tree 25.1–50 m	> 33″ DBH
	Late-Closed	41–100%	Tree 25.1–50 m	> 33″ DBH
East Cascades Mesic Montane Mixed Conifer Forest and Woodland (Grand Fir series)	Early	0–100%	Tree 0–10 m	Sapling >4.5′, <5″ DBH
	Mid-Open	0–60%	Tree 10.1–25 m	9–21″ DBH
	Mid-Closed	61–100%	Tree 10.1–25 m	9–21″ DBH
	Late-Open	0–60%	Tree 25.1–>50 m	>33″ DBH
	Late-Closed	61–100%	Tree 25.1–>50 m	>33″ DBH
Rocky Mountain Subalpine Dry-Mesic Spruce-Fir Forest and Woodland (Subalpine Fir series)	Early	0–40%	Shrub 0–0.5 m	None
	Mid-Open	11–30%	Tree 5.1–10 m	9–21″ DBH
	Mid-Closed	31–60%	Tree 5.1–10 m	9–21″ DBH
	Late-Open	11–40%	Tree 10.1–25 m	21–33″ DBH
	Late-Closed	41–70%	Tree 10.1–25 m	21–33″ DBH

2.7. Limitations of Methods

As we did not have pre-fire vegetative data, and the Camp et al. [11] plots were not monumented, we inferred pre-fire vegetation attributes with only the burned post-fire vegetation available to sample, which is standard for the CBI protocol [49]. Vegetative reconstruction estimates can be problematic due to the uncertainties of differentiating the magnitude of observed effects solely due to the fire event from the pre-fire conditions and other ecological changes (such as an insect attack or hydrological flow) that occur between the fire and the post-fire measurements [62–64]. Canopy cover is particularly difficult to measure from ocular estimates [65], especially after the canopy has been partially consumed in a fire, and it is reasonable to assume that the two-year lag between the fire and our field data collection introduced further error. However, we felt that the opportunity to revisit forested stands that were classified for a refugial objective in a prior study in order to describe fire effects was an opportunity that could not be ignored, despite these limitations.

3. Results

3.1. Assessment of Fire Effects in the Camp et al., Refugial and Non-Refugial Plots

Of the 109 paired plots that were established initially by Camp et al. [11] and that we revisited after the 2012 wildfires, only six (5.5%) did not experience any fire effects in 2012 (i.e., they were unburned). Only one of these six unburned plots was classified as a fire refugium by Camp et al. [11] (Table 5), although this difference in burn occurrence between refugial and non-refugial plots was not statistically significant (chi-square test of independence, $\chi^2 = 0.768$, $p = 0.381$) due to the low number of samples. This translates to <1 percent of plots persisting as refugia and 4.5% having no fire effects and becoming refugia under the most conservative definition.

Table 5. Number of plots classified in the original Camp et al. [11] study that burned in the 2012 fires, re-surveyed in the present study, paired to the plots from [11], and found to have no fire effects (i.e., the most conservative definition of refugial) by this study.

Sampling Description	Refugial per [11]	Non-Refugial per [11]	Total
Sampled by [11]	43	183	226
Sampled in 2014, this study	41	81	122
Plots paired to [11]	36	73	109
Plots unburned	1	5	6

A comparison of the fire effects between the 109 matched Camp et al. [11] refugial and non-refugial plots revealed a trend in differences in burn severity, where classified refugial plots generally experienced greater fire effects (Figure 4). Refugial plots burned more severely than non-refugial plots as assessed through the total plot CBI metric (W = 1015.5, p = 0.0582). For the tree-level severity metrics, the 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 significantly higher for refugial plots (at α = 0.1, used due to non-normal data and a greater number of non-refugial plots). Percent overstory tree mortality and percent bole char observations were also higher for refugial plots, but these differences were 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.

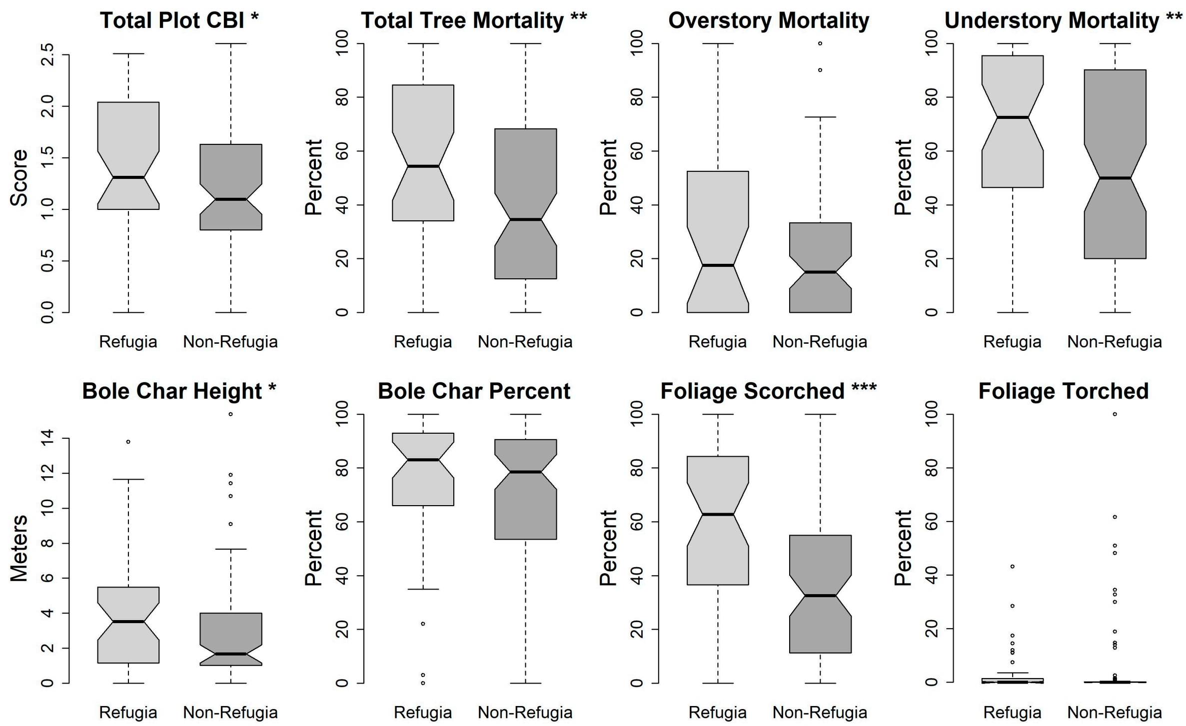

Figure 4. Comparison of fire effects between Camp et al. [11] refugial and non-refugial plots for eight different burn severity metrics. * $p \leq 0.10$, ** $p \leq 0.05$, *** $p \leq 0.01$.

3.2. Changes in Successional State

Based on our reconstruction of forest structure and successional state pre-fire, there were no Early Development successional state plots pre-fire, with most of the plots classified at Mid-Development

(32% Open Canopy and 39% Closed Canopy), and the remaining 29% classified as Late Development (9% Open Canopy and 20% Closed Canopy) (Table 6). Post-fire, 19% of sample plots were classified as an Early Development successional state, 26% were classified as Late Development, and most plots were still Mid-Development (55%). Half (50%) of the sampled plots did not change in terms of the successional state due to fire effects, while nearly one-third (31%) transitioned from a closed to open-canopy structure (for both Mid- and Late Development categories combined). Closed Canopy plots classified as either Mid- or Late Development successional state (59%) were more abundant than Open Canopy plots pre-fire (41%), but Open Canopy plots (65%) were four times more abundant than Closed Canopy plots (16%) post-fire.

Table 6. Distribution of successional states for all 122 sampled plots according to pre-fire and post-fire successional state for all Biophysical Setting models.

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

Assessing structural transitions by whether plots were classified as refugial or non-refugial by Camp et al. [11] reveals that a higher proportion of pre-fire refugia transitioned to Early development successional state (24%) than non-refugial plots (16%) (Table 7). More refugia were also thinned by the fire from Closed to Open Canopy (34%; in both Mid and Late-development states combined) than non-refugia (30%). Accordingly, more non-refugial plots maintained their pre-fire successional state.

Table 7. Primary plot structural transitions by pre-fire refugia classification.

Successional Transition	Refugia		Non-Refugia	
	Count	Percent	Count	Percent
Maintained state	17	42%	44	54%
Thinned from Closed to Open Canopy	14	34%	24	30%
Converted to Early Development state	10	24%	13	16%

4. Discussion

4.1. Comparison of Fire Effects between Sample Years

We found that the plots classified by Camp et al. [11] as refugial experienced more severe fire effects from the 2012 wildfires in comparison to classified non-refugial plots. This finding supports an inference of Camp et al. [11] in their study; they noted that pre-European settlement fire refugia appeared to have higher fire intensities and severities associated with longer fire return intervals. What was particularly surprising in the present study was just how few of the 122 plots sampled were unburned (<6%). Studies quantifying the unburned proportion across entire fires at local to regional scales have found a very broad range of the proportion unburned for individual fires [4,42], with Meddens et al. [13] reporting a regional average of 20% unburned for the inland Northwest. As these plots were specifically located by Camp et al. [11] to capture prospective fire refugia, the low proportion of unburned compared to regional averages further supports their conclusion that refugial plots, when they do eventually burn, do so with a higher severity than the non-refugial surrounding matrix. Because of the longer fire return intervals in refugial patches, tree species with thin bark and minimal self-pruning, such as grand fir, are able to establish and develop as ladder fuels that facilitate the surface fire ignition of crown fuels due to their lower canopy base height [32], a concept that Camp et al. [11] termed 'outgrowing' the refugial status. The additional fuels accumulated in these 'outgrown' refugial patches then lead such sites to burn at a higher severity than the surrounding matrix.

4.2. Distribution of Post-Fire Successional States and Ecological Transitions

We found that less than 30 percent of plots resampled in 2014 were Late Development stage pre-fire, with Mid-Development stage plots being the most common. This successional composition is relatively consistent with the pre-European settlement fire regime [39], although occasional high severity patches of fire (as part of the mixed-fire regime) would have also produced a few Early Development patches scattered across the landscape during that period. Here, however, we encountered no plots that were Early Development successional state pre-fire, which was unsurprising given that prior to 2012, the most recent wildfire in the study area occurred before 1900 due to the effectiveness of US fire suppression policies in the 20th century [46]. Post-fire, 19 percent of sampled plots transitioned to the Early Development state, increasing the structural heterogeneity of the study area [66]. Open Canopy stands were four times more abundant than Closed Canopy stands post-fire, whereas Closed Canopy stands were more abundant pre-fire. These sorts of transitions are consistent with the restoration needs highlighted by Haugo et al. [59] for these forests, and suggest that in this case study, at least, the fire behavior may have played a restorative role by creating both early successional openings and a more open canopy from a previously closed canopy.

The relationship between the more severe fire effects and the forest structural changes becomes evident when stratifying structural changes by whether plots were classified as refugia pre-fire. This fire-induced transition to an increased open canopy structure is consistent with mixed severity fires reducing canopy closure [32,43,67], and occurred in a higher proportion of refugial plots because they burned at a higher severity. Similarly, more refugial plots transitioned to Early Development successional state due to a higher burn severity in refugial plots. One point of interest in this breakdown is the number of plots that maintained their successional state (as there were no Early state plots pre-fire, these were all Mid- and Late-Development sites). Most of these plots (all but six) experienced some fire effects; these effects were of a low enough severity to maintain the forest structure successional state given the definitions in the BpS models.

4.3. Implications for Management

In the face of environmental change, managers can take many actions to increase ecosystem resilience [36,68]. Many plant and animal species are fire-sensitive and require refugial habitats to persist in the landscape [2], including species of high management concern such as the northern spotted owl (*Strix occidentalis caurina*), which was the focal species of the Camp et al. [11] refugia analysis. There is much concern over the persistence of refugia given climate change and changing ecological disturbance regimes [41], but our results demonstrate that even with fire re-entry into sites with over a century of fire exclusion, some refugial plots still maintained pre-fire forest structure and some previously non-refugial plots served as during-fire refugia in the 2012 fires. For managers seeking to conserve refugia for given species, there are likely pathways to doing so that do not require the total exclusion of wildfire, but additional research is needed to improve quantitative landscape composition models and determine when forests have become vulnerable to permanent structural transitions that can eradicate refugia [69].

5. Conclusions

Wildfire refugia in forested landscapes are critical to species survivorship both during and between fires, as well as facilitating forest regeneration in adjacent burned areas. However, refugia research is still nascent, with the definitions and delineation of refugia being highly variable and dependent upon species of interest. This case study provided a unique opportunity to assess the effects of a wildfire on fire refugia in central Washington State, USA, that were classified and delineated by Camp et al. [11]

over two decades ago and subsequently burned in two 2012 wildfires. Our findings that almost all of the plots burned, but half of the plots persisted in their pre-fire forest structure successional state, suggests that definitions of fire refugia focused on the maintenance of forest structure or canopy thresholds may reveal a higher proportion of refugia that are persistent through multiple fires. We also found that nearly one-fifth of the plots converted to Early Development state due to a high burn severity, and almost one-third experienced enough crown loss to transition from Closed to Open Canopy; these types of transitions are critical to forest restoration following over a century of fire exclusion in forested landscapes across the western US. Ultimately, this opportunistic case study highlights that the re-entry of fire into forests where fire has been excluded produces a range of fire effects that include the maintenance of pre-fire forest structure. Additionally, many current fire refugia may be vulnerable to future fire, particularly where fire exclusion has allowed for fuel accumulation and the growth of fire-sensitive species. Refugia more broadly, however, require much additional study to improve our understanding of their persistence, vulnerability, and role in forest structural dynamics.

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