FINAL REPORT

Title: Effects of post-fire management on vegetation and fuels following successive wildfires in mixed conifer forests

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List of Abbreviations

ASL – above sea level
DBH - diameter at breast height
FACTS - Forest Service Activity Tracking System
FTI - Fire Tolerance Index
GAM - Generalized Additive Model
GLM - Generalized Linear Model
Ha - hectare
LiDAR - Light Detection and Ranging
Mg – megagrams
MSE – Mean squared error
RdNBR - Relative differenced Normalized Burn Ratio
STI - Shade Tolerance Index
TWI - Topographic Wetness Index

Keywords

reburn, interacting fires, high-severity fire, fuel dynamics, forest resilience, vegetation dynamics, post-fire restoration

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Abstract

In the face of changing climatic regimes and increases in extreme fire events, many western forests are poised to burn, not only once but multiple times, sometimes in short succession. As such, land managers have limited opportunities to effectively alter post-fire vegetation and fuels to make them more resilient to future disturbances like fire. In this study, we took advantage of a unique opportunity to examine vegetation and fuels development after successive fires, and in doing so identified several management-relevant pathways by which post-fire vegetation structure and fuels influence the severity and ecological outcomes of a subsequent wildfire.

We analyzed three datasets, collected at multiple spatial and temporal scales, from three overlapping fires in northern California. First, we collected and analyzed data from 134 field plots, established after an initial mixed severity fire and remeasured after a subsequent reburn. Second, we used repeat LiDAR data, collected over the same time period, to expand our analysis to a larger spatial scale. Finally, we used high-resolution aerial orthoimagery to assess how post-fire vegetation and fuel development influence fire severity patterns in a reburn.

Our results suggest that resistance to high-severity reburn is contingent on a combination of factors, including topography, fire weather, vegetation structure, and woody fuels. In our study fires, areas with more variable and mesic terrain were less likely to reburn at high severity. In stands that burned at lower initial severities, high relative humidity and low wind speeds during the reburn also reduced the likelihood of high-severity fire effects. In areas that initially burned at high severity, high densities of snags and down woody fuels were associated with high-severity effects in the second fire. Variability and density of vegetation also played an important role in moderating reburn severity. In early-seral habitats (i.e. those that burned initially at high severity), areas with relatively sparse understory and variable sub-canopy were most likely to avoid repeat high-severity fire. Forests that burned initially at low to moderate severity were most likely to resist high-severity fire if they supported sparse understories and relatively dense but heterogeneous vegetation in the upper strata (> 2 m in height).

Following the reburn, areas impacted by successive high-severity fires had little to no live conifer overstory, retained high cover of shrubs in the understory, and had little to no conifer regeneration. In contrast, successive low to moderate severity fires significantly reduced tree density, increased tree regeneration, and in some severity combinations (e.g. low followed by moderate severity) promoted colonization by shrubs. In our study, multiple low to moderate severity fires did not shift forest composition toward dominance of more fire tolerant or shade intolerant species.

Taken together, the results of our study suggest that post-fire vegetation structure and woody fuels play an important role in subsequent fire severity patterns and ultimately influence the resilience of post-fire landscapes to future fire. In areas where high-severity reburn is undesirable, managers should consider treatments that reduce the density and continuity of vegetation, standing snags, and large woody surface fuels. In areas where proactive reforestation is necessary, planting in areas that are in rough or mesic terrain may reduce the likelihood of high-severity reburn. The results of our study also suggest that active post-fire management may be necessary in areas that have burned at low to moderate severity in order to maintain or promote the restorative benefits of an initial fire or to restore the dominance of fire resilient tree species.

Objectives

The purpose of our study was to examine how first-entry wildfires affect vegetation structure, composition, and fuel loads, and to assess how these variables influence the severity and ecological outcomes of a subsequent wildfire (Figure 1). To address this goal, we utilized a unique set of repeat measurements, collected at multiple spatial scales, in three overlapping mixed-severity fires in northern California. These data allowed us to analyze the wide range of conditions found in burned landscapes, including those most often targeted by managers in post-fire restoration efforts.

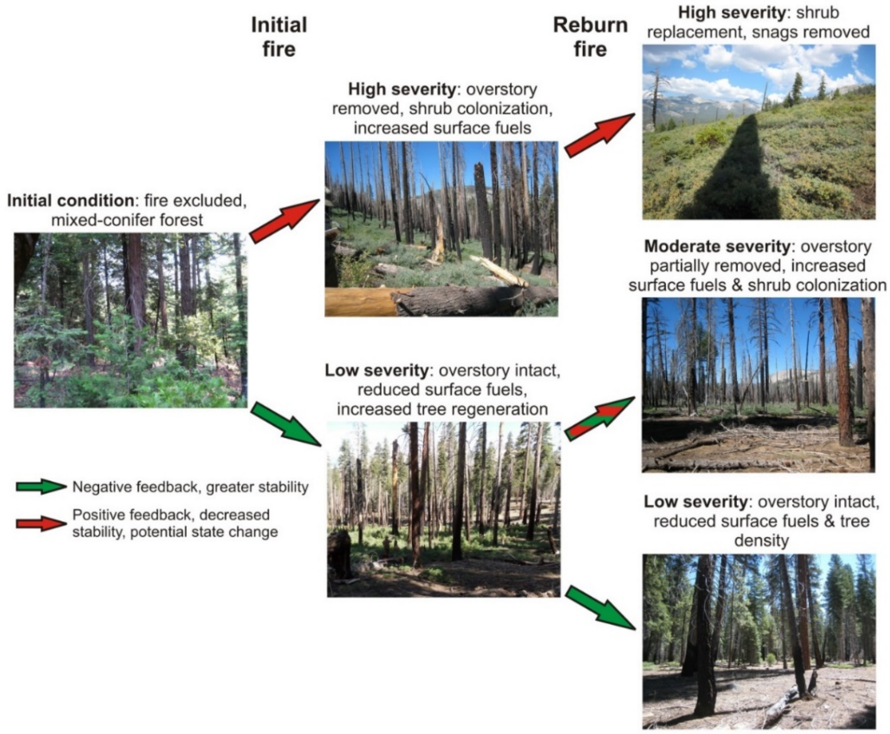

Figure 1. Conceptual model of potential pathways (hypotheses) for post-fire vegetation and fuel dynamics following initial fires and short interval (5-15 year) reburns. Pathways are coded for the type of ecological feedback that we expected would occur in response to different fire severity levels and the effects these changes would have on subsequent fires.

Objective 1: Examine post-reburn vegetation and fuel loadings to test hypotheses about future successional trajectories under potential positive and negative feedback loops (see Figure 1). We tested the following two hypotheses:

(H1) In montane mixed conifer forests affected by a century of fire suppression, successive high-severity fires create a positive feedback that promotes dominance by shrubs, and homogenization of vegetation structure and diversity at the landscape scale.

(H2) Multiple low to moderate intensity fires in fire-excluded montane mixed conifer forests will result in a stabilizing negative feedback by reducing surface fuels and small tree density, while maintaining larger overstory trees and promoting landscape-scale forest heterogeneity and structural diversity.

We met this objective by collecting and analyzing data at two different spatial scales. First, we compared vegetation and fuel measurements from 134 field plots (0.04 ha) that were established prior to the reburn and resampled as part of this study. Second, we used LiDAR (Light Detection and Ranging) data, collected after the initial 2000 Storrie Fire and after the 2012 reburn, to investigate changes in forest structure across a range of severities at larger spatial scales.

Objective 2: Determine if post-fire management activities (e.g. tree planting, thinning, salvage logging, and fuel reduction) can significantly affect fire severity and influence plot and landscape-scale vegetation patterns and fuel loadings after subsequent reburns. Our original intent was to test the following two hypotheses using a combination of field measurements and LiDAR data:

(H3) Post-fire management activities that reduce standing snags, tree density, and surface fuels (e.g. fuel reduction activities and salvage logging) can shift the fire regime towards low to moderate severity fires and promote patterns consistent with H2 above.

(H4) Post-fire management activities that increase fuel pools (e.g. tree planting and salvage logging with no fuel reduction component) after an initial fire will promote subsequent stand-replacing fires and type conversion to shrub-dominated systems, thus reducing heterogeneity and structural diversity at the landscape scale.

This objective was partially met. In 2016 we identified over 647 ha within our study area that had a record of post-fire management in the Forest Service Activity Tracking System (FACTs) database (USDA Forest Service 2016). These treatments included salvage logging, thinning, fuel reduction, and tree planting. Due to the small number of previously established plots within treatment units, we established an additional 24 field plots in 2018 to investigate the relationship between post-fire management activities (implemented after the initial fire, but prior to the reburn) and trajectories of vegetation and fuel loadings after a subsequent reburn. We also conducted preliminary analyses using two LiDAR datasets, one acquired after initial post-fire treatment implementation and one acquired after the subsequent reburn, to investigate changes in vegetation cover (at different height strata) in treated and untreated areas. Unfortunately, field reconnaissance of the areas treated after the initial fire (identified in Table 1 of our proposal) revealed significant discrepancies between on-the-ground treatments and those recorded in the FACTs database. Many of the treatments originally included in our analysis turned out to be unconventional (i.e. not representative of typical salvage or planting efforts; see Figure 2) and several units were treated multiple times, including after initial field data collection and after the reburn. These factors significantly limited the potential for replication, and ultimately limited our ability to examine the effects of post-fire management actions on reburn severity and post-reburn succession.

Figure 2. Example of an unconventional post-fire reforestation treatment , where tree seedlings (identified by small cages in photograph) were planted in the understory of an intact stand that burned at low to moderate severity.

Because we were unable to directly assess the effect of post-fire management, we focused our analysis on attributes of the post-fire landscape that managers commonly target in post-fire restoration treatments - fuels and forest structure. We used high-resolution imagery collected nine years after the 2000 Storrie Fire to measure snags, logs, and shrub cover in areas impacted by high-severity fire, and to assess how fuel development after this initial fire influenced reburn severity in the subsequent wildfire. We also used LiDAR data, collected concurrently with the aerial imagery after the Storrie Fire, to evaluate how variability in vegetation structure and density influenced severity and ecological outcomes of the second fire.

<u>Additional challenges to meeting objectives</u>

Regional budget issues in 2017 resulted in a 25% reduction in the total amount of funding we received and significantly impacted our ability to directly compensate Forest Service personnel for work related to this JFSP project. As described to JFSP, these constraints reduced funding for our 2017 field crew, created the need to complete LiDAR analyses with non-USFS personnel, and eliminated our ability to host a large workshop for managers. These challenges, combined with the inconsistencies associated with post-fire treatments (described above), resulted in a narrower focus for Objective 2.

Background

Increasing incidence of large wildfires with extensive high severity effects across the western U.S. has shifted the attention of land managers towards post-fire landscapes. Restoration of specific resources damaged by wildfire has traditionally been the focus of management efforts; however there is growing recognition that larger scale post-fire management may be needed to promote resilience to future reburns, increase biodiversity, and reestablish pre-fire vegetation types, such as montane mixed conifer forests.

Fires consume fuel, which can limit the extent and effects of subsequent fires (Collins et al. 2009, Parks et al. 2014). This type of feedback pattern, where low to moderate severity fires consume surface and in some cases ladder fuels and reduce the intensity and effects of subsequent fires (Figure 1), exists in some contemporary forests (e.g. Collins et al. 2009, Larson et al. 2013). However, this is not the case for much of the dry forest types in the western U.S., where decades of fire exclusion and overstory removal harvesting have altered forest structure and contemporary fire patterns.

In places where post-fire tree mortality is high, there is increased potential for hazardous surface fuel accumulations, as fire-killed trees lose limbs and ultimately fall (Ritchie et al. 2013). In addition to elevated fine and coarse woody debris, there can be a strong shrub response after fires (Russell et al. 1998, Collins and Roller 2013), which can further add to hazardous surface fuel conditions (Albini 1976). There is increasing concern that these hazardous fuel conditions (i.e., standing snags, down wood, and dense shrub vegetation) will set forests on a positive feedback trajectory with subsequent fires; one in which extensive stand-replacing fire will promote more stand-replacing fire (Thompson and Spies 2010, Coppoletta et al. 2016). This type of feedback has the potential for long-term state change from forest to persistent shrubland or grassland (Collins and Skinner 2014). Climate change may be increasing the likelihood of these long-term fire and vegetation state changes (e.g. Westerling et al. 2011, Larson et al. 2013).

Land management in the western U.S. is at a critical junction. In the face of changing climatic regimes and increases in extreme fire events, many dry forests in the western U.S. are poised to burn, not only once but multiple times, sometimes in short succession. Managers may have a limited window of opportunity to effectively alter post-fire trajectories. Therefore, it is even more crucial to determine how to alter post-fire conditions so that disturbance processes (like wildfire) can act to increase rather than reduce forest heterogeneity and provide landscapes with the ecological flexibility to persist through future climatic changes.

Previous work conducted by many of the Co-PIs on this study demonstrated that high-severity fire effects in an initial fire can create vegetation and fuels conditions that promote high-severity fire effects during a subsequent reburn (Coppoletta et al. 2016). In this JFSP study we went beyond explaining drivers of reburn severity in three unique ways:

(1) We collected data in 134 permanent field plots that were established after an initial mixed severity fire and remeasured after a subsequent reburn, to gain insight into changes in vegetation and fuels after successive fires.

(2) We analyzed repeat high-resolution airborne LiDAR data, collected after the initial fire and after the reburn, to expand this plot-level analysis to larger spatial scales.

(3) We used high-resolution aerial orthoimagery collected nine years after the initial fire to measure vegetation and fuel loads within high-severity patches, and to assess how fuel development influenced fire severity in the subsequent reburn.

Methods

Study area

Our study area is located on the Plumas and Lassen National Forests in the northern Sierra Nevada mountains of California. It encompasses three fires that burned over a 12-year period: two initial fires (the 2000 Storrie Fire and the 2008 Rich Fire), and one reburn (the 2012 Chips Fire). The topography of the area is generally mountainous and steep, with elevations ranging from 673 to 2144 meters above sea level. Climate in the region is characterized as Mediterranean, with most precipitation falling as rain or snow during the cold winter months, followed by hot and dry summers.

Prior to the initial fires, vegetation in our study area was predominantly lower montane mixed-conifer forest, with approximately 95% of the stands classified as having moderate to dense overstory canopy (CALVEG 2004). Dominant species included white fir (*Abies concolor*), Douglas fir (*Pseudotsuga menziesii*), ponderosa pine (*Pinus ponderosa*), Jeffrey pine (*Pinus jeffreyi*), sugar pine (*Pinus lambertiana*), incense-cedar (*Calocedrus decurrens*), and California black oak (*Quercus kelloggii*). Above 1800 m elevation, mixed-conifer forest transitioned to white fir and red fir (*Abies magnifica*) forest. Shrublands, characterized by *Ceanothus* and *Arctostaphylos* (manzanita) species, dominated lower elevation, drier, and southeast-facing canyon slopes. Although forests in the area experienced a frequent, low- to moderate-severity fire regime historically (mean fire return interval of 8 to 22 years; Moody et al. 2006), less than 15 percent of the study area had experienced fire in the century preceding the initial fires (California Department of Forestry and Fire Protection 2020).

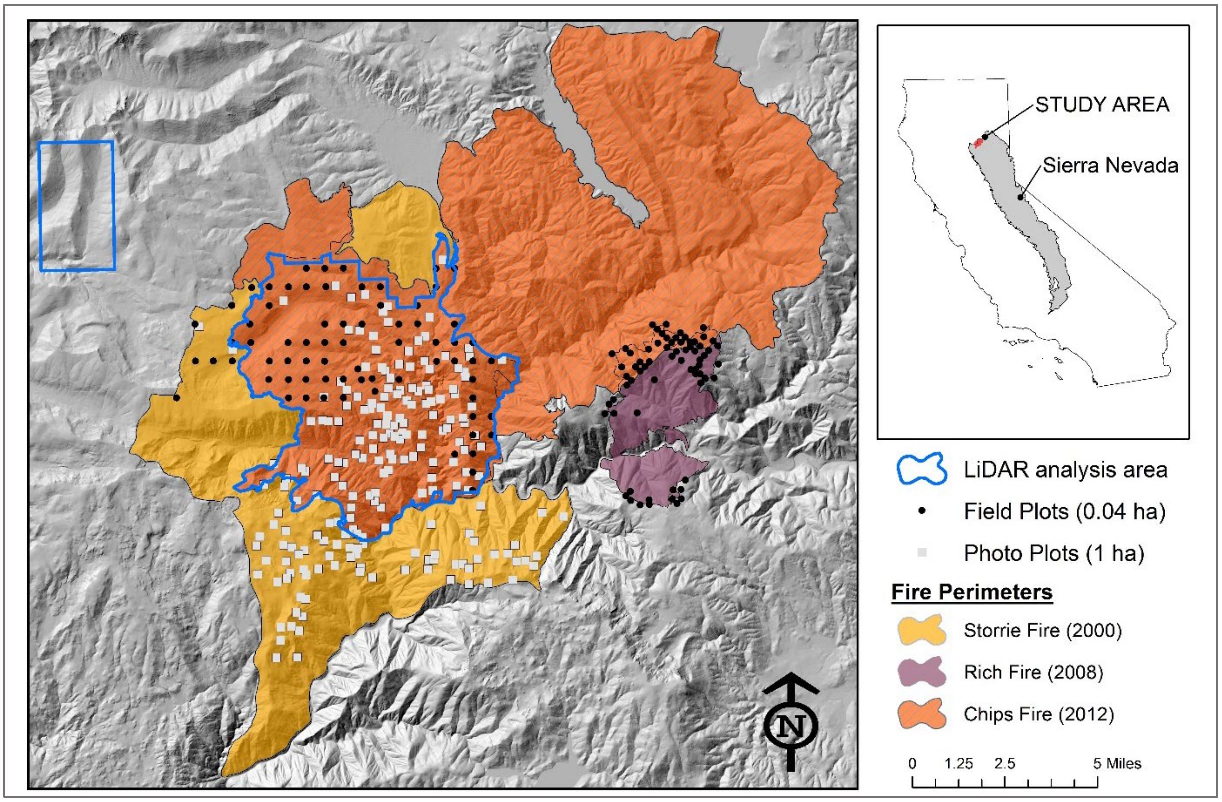

Figure 3. Map of study area showing field plots, LiDAR data analysis area, and orthoimagery plot locations.

Data collection and analysis

We used three different analytical approaches to address our study objectives; each assessment is described separately below.

Field plots

To examine post-reburn vegetation and fuel loadings (Objective 1), we resampled 134 common stand exam field plots that were established after the 2000 Storrie Fire (n=68) and 2008 Rich Fire (n=66) and were subsequently reburned by the 2012 Chips Fire (Figure 3). Plot centers were placed on the vertices of an 800-m grid in the Storrie Fire and a 200-m grid in the Rich Fire. Fire severity for each plot was categorized based on the Relative differenced Normalized Burn Ratio (RdNBR; Miller and Thode 2007).

Table 1. The number of plots sampled within each fire severity combination.

	Reburn severity[1]				
Initial severity[1]	**Unburned**	**Unchanged**	**Low**	**Moderate**	**High**
Unburned	14		5	2	1
Unchanged		2	6	3	
Low	13	8	18	11	1
Moderate	11	5	4	4	2
High	7	4	2	7	4

[1] Severities were defined as follows: unburned: outside fire perimeters; unchanged: within a fire perimeter and RdNBR < 69; low: RdNBR = 69-315; moderate: RdNBR = 316-640; high: RdNBR ≥ 641

Figure 4. Example of plots burned: (a) twice at low severity; (b) high then low severity; and (c) twice at high severity

Sampling was conducted over two field seasons (2017 and 2018) and was consistent with methods initially used to measure these plots. Field data were collected using the USDA Forest Service common stand exam protocol, which consists of two concentric circular plots originating from a single plot center (USDA Forest Service 2009). A larger plot was used to collect tree data (>12.7 cm diameter at breast height, dbh), including species, status (live or dead), dbh, height, and height to live crown; sprouting response was also categorized for all oaks. In the Rich Fire, tree data were collected in a large plot with a fixed 11.3-m radius, whereas the larger plots in the Storrie Fire employed a variable radius design. Sampling in both fires utilized a 5-m radius smaller circular plot where the number, species, dbh, height, and status of saplings (> 1.7 m tall) and seedlings (< 1.7 m tall) was recorded. Ocular estimates of tree, shrub, forb, and graminoid cover were conducted within the 5-m fixed radius plot. Ground cover data were also collected within the small plot for the following categories: rock, bare soil, wood (> 3” diameter), litter and duff, and basal vegetation. Fuels data were collected along four transects per plot using a planar intersect technique (Brown 1974).

Topographic variables for each plot were derived using a combination of GIS (elevation) and field measurements (slope and aspect). Distance to closest seed source was obtained by overlaying plot GPS locations with post-fire aerial imagery and measuring the distance from the plot to the closet standing live tree. Information about treatment activities were obtained from the FACTs database (USDA Forest Service 2016) and evidence of treatment was verified for each plot in the field. Sample years since last fire and years between fires were derived by calculating the number of years between data collection and the most recent fire, and if reburned, the years between the initial and reburn fires for each plot, respectively. The presence of serpentine soil was also recorded based on the California soils map developed by the National Cooperative Soil Survey.

Plot-level shade and fire tolerance indices were derived for plots pre- and post-reburn, using canopy tree importance values weighted by tolerance rankings derived from existing information in the scientific literature (Table 2; Evans et al., 2011; Hallin, 1957; Minore, 1979). Higher tolerance values correspond with relatively greater shade and fire tolerance. The importance value for each species (IV_i; Curtis and McIntosh, 1951) was calculated as a measure of the dominance of each species at the plot level:

$$\mathrm{IV}_i = \left((BA_i \div BA_{k\ total}) \times 100\right) + \left((D_i \div D_{k\ total}) \times 100\right)$$

where BA_i was the basal area ($m^2\ ha^{-1}$) and D_i the density ($trees\ ha^{-1}$) of species i, and $BA_{k\ total}$

and $D_{k\,total}$ were the total basal area and total density of all species in plot *k*, respectively.

A plot-level Shade (STI) and Fire (FTI) tolerance index value was then calculated using the following equation:

$$STI\ or\ FTI = \sum IV_i \times TR_i$$

where TR_i was the species-specific shade or fire tolerance ranking (Table 2).

Table 2. Fire Tolerance Index (FTI) and Shade Tolerance Index (STI) rankings for mixed conifer tree species in our study plots. Higher rank values for FTI and STI correspond with relatively greater fire and shade tolerance.

Species	Common Name	FTI	STI
Abies concolor (ABCO)	White fir	2	8
Abies magnifica (ABMA)	Red fir	1	7
Calodedrus decurrens (CADE)	Incense cedar	2	4
Pinus contorta (PICO)	Lodgepole pine	1	3
Pinus jeffreyi (PIJE)	Jeffrey pine	5	2
Pinus lambertiana (PILA)	Sugar pine	3	5
Pinus ponderosa (PIPO)	Ponderosa pine	6	1
Pseudotsuga menziesii (PSME)	Douglas fir	4	6

Field Data Analysis

For our initial analysis, we focused on five response variables: live tree density (ha^{-1}), seedling density (ha^{-1}), shrub cover (%), FTI, and STI. These variables did not meet the assumptions of normality even after data transformation. Therefore, we used a Wilcoxon signed-rank test, a non-parametric test, to test for significant differences between pre- and post-Chips Fire samples for all variables in all plots (n = 134). The three variables related to live trees (e.g. tree density, FTI, and STI) were also tested using only plots with live trees after the Chips Fire reburn (n = 52). Species-wise means of importance values were calculated and graphed, including both pre- and post-Chips Fire samples and based on the interaction of unchanged, low, and moderate initial and reburn severities.

The response variables used in the regression analyses were calculated based on the plot-level change in the variable between the two sampling periods (i.e. the difference between pre- and post-Chips Fire samples). Response variables included: change in density of live trees and seedlings (all species combined); shrub cover; and STI and FTI (Table 3). The statistical analysis procedure was a multi-step process that first selected the most important explanatory variables to avoid regression model overfitting (Table 3). A total of 19 explanatory variables were included in the random forest models. Explanatory variables were identified for inclusion in the initial model selection process based on published literature, including physical and vegetation variables with some effect on the same or similar response variables in previous analyses (i.e., Collins et al., 2007, Coppoletta et al., 2016). Model selection was conducted using random forest analysis with the randomForest package (Liaw and Wiener, 2002) in R (R Core Team, 2019). The explanatory variables with the greatest importance (% increase in MSE) from each random forest model were selected for inclusion in subsequent Generalized Linear Models (GLM) conducted in R.

Table 3. Importance values obtained from the random forest models. Response variables are the change (Δ) from pre- to post-reburn in tree density, FTI, STI, seedling density and shrub cover. The non-severity explanatory variable with the greatest importance for each model is indicated in bold.

Explanatory variables	Δ Trees density ha^{-1}	Δ FTI ha^{-1}	Δ STI ha^{-1}	Δ Seedling density ha^{-1}	Δ Shrub cover ha^{-1}
Chips Severity (0-5)	27.674	26.198	19.169	2.146	6.691
Initial Severity (0-5)	13.339	15.363	18.967	14.235	16.046
Slope (°)	-0.012	2.334	5.512	7.677	4.661
Aspect (°)	14.421	3.407	-7.485	4.210	1.724
Elevation (m asl)	3.635	-15.719	-3.682	18.575	8.366
Sample years since last fire (count)	20.099	**14.963**	**21.079**	4.590	11.051
Years between fires (count)	12.495	8.298	7.457	12.295	12.214
Number fires (count)	16.537	6.996	7.295	3.641	7.757
Reburn (1/0)	12.266	8.431	5.681	1.830	7.093
Serpentine lithology (1/0)	2.218	3.128	7.057	6.511	4.944
Treatment (1/0)	4.700	-0.249	1.151	9.572	0.194
Post-Chips rock (%)	4.173	0.554	2.615	5.549	-0.832
Pre-Chips coarse woody debris (mg ha^{-1})	-3.250	-1.593	-3.826	3.178	3.354
Post-Chips coarse woody debris (mg ha^{-1})	5.583	-6.792	-5.892	1.092	14.490
Pre-Chips fine woody debris (mg ha^{-1})	16.162	7.172	-1.440	3.594	4.724
Post-Chips fine woody debris (mg ha^{-1})	13.536	7.868	4.111	-0.281	9.075
Pre-Chips shrub cover (%)	**22.548**	7.235	12.697	9.458	
Post-Chips shrub cover (%)	8.138	13.363	12.124	**19.778**	
Pre-Chips mixed conifer dead trees ha^{-1}	-0.705	-1.251	-0.137		
Distance from closest live seed source (m)				7.783	**16.477**
Post-Chips mixed conifer live trees ha^{-1}				12.082	14.039
Pre-Chips mixed conifer live trees ha^{-1}				9.772	10.202

The base GLM models for each response variable included a categorical variable for initial fire severity (i.e., Storrie or Rich) and reburn severity (Chips Fire), as well an interaction between initial and reburn severity. Additional explanatory variables were added to the models, starting with the non-severity variable with the greatest importance value from the random forest analysis. Explanatory variables were included in the GLM if the effect on the dependent variable was significant ($\alpha = 0.05$) and the process of adding variables was stopped once the next most important variable did not have a significant effect. The additional explanatory variable included in the tree density GLM (pre-Chips Fire shrub cover) was centered on its mean of 25%.

Response variables were not normally distributed and were right skewed. Since response variables included both positive (i.e., increases from pre- to post-Chips Fire samples) as well as negative values (i.e., decreases between samples), response variables were transformed by adding the most negative value plus one to all observations. The GLMs were specified using the Gamma distribution family and log link function. Post-hoc analyses of residuals and outlier tests were performed, and significant outliers greater than two-standard deviations from the mean were removed before final model runs. Reductions in Akaike Information Criterium (Δ AIC >2), and residual deviance were also used to determine whether the additions of variables to the base model and removal of outliers improved the models. Post-GLM margins analysis was employed to examine the interaction effects of initial and reburn severities on each response variable, while holding other explanatory variables at their means.

LiDAR data

We utilized two high-resolution LiDAR datasets to address Objectives 1 and 2. The first LiDAR data were collected in 2009 and covered the entire Storrie Fire footprint. The second were collected three years after the reburn in 2015 and included portions of the Storrie and Chips fires as well as unburned watersheds to the north. For this analysis, we focused on the area of overlap between the 2000 Storrie and 2012 Chips fires, as well as an adjacent unburned reference site (1774 ha) located approximately 10 km from the fire perimeters (Figure 3).

LiDAR data were used to measure vegetation area density (VAD) across three vertical strata: 1-2 m above ground height (understory), 2-8 m (sub-canopy), and 8-80 m (canopy). VAD was calculated using the lidR package (Roussel and Auty 2019). We used a corrected grid-based version of the methods described in Bouvier et al. (2015):

$$P_i = \frac{N_{[0;z]}}{N_{[0;z+dz]}}$$

Where P_i is the gap fraction of the focal stratum (used to calculate VAD), z is the lower bound of the current stratum, and dz is the depth of the focal stratum. $N_{[0;z]}$ is the number of returns from heights below z, and $N_{[0;z+dz]}$ is the number of returns from heights below $z + dz$. Thus, gap fraction is estimated as the proportion of LiDAR returns that passed through the focal stratum. This calculation included all pulse echoes (not just first returns). We used a grid resolution of 10-m as the finest grid size, which gave valid VAD estimates for all strata (at least 1 return below 1 m height per 10m^2 pixel even in the densest vegetation) over >= 99.999% of the study area. VAD values were log transformed for all analyses and are subsequently referred to as "vegetation density".

Elevation, topographic roughness, and topographic wetness index (TWI) were calculated using a 30-m DEM and the elavatr package (Hollister and Shah 2017). Roughess was calculated as the standard deviation of elevation within a focal area. TWI was calculated at 90-m resolution using the dynatopmodel package (Metcalfe et al. 2018). We considered incorporating additional topographic variables (e.g. southwestness and topographic position index) but dropped them from the final analysis because they did not improve out-of-sample predictive performance of preliminary models. Daily fire weather data (~ 4 km resolution) from the gridMET database (Abatzoglou 2013) were combined with fire growth polygons from the GeoMAC database (National Interagency Fire Center 2019) to estimate mean minimum relative humidity and mean daily windspeed for each sample point on the day it burned in the Chips Fire. We used Landsatderived fire severity estimates (RdNBR) that were subsequently classified into percent canopy cover mortality (Miller and Quayle 2015) to delineate the cut-off (75% canopy mortality) between low/moderate- and high-severity fire effects.

LiDAR Analysis

We generated a 200-m point sampling grid across both the reburn and unburned reference areas. Sample points were limited to areas that were lower-montane conifer forest prior to both fires (CALVEG 2004). We also excluded areas that experienced management (USDA Forest Service 2016) between the 2000 Storrie Fire and the relevant LiDAR sampling year (2009 or 2015). At each sample point (n = 2099) we extracted topographic variables, Chips Fire weather data, and pre-reburn vegetation density using the 2009 LiDAR data. We summarized environmental data at various scales using the following radii around each point: 25, 50, 75, 100, 125 and 150 m. To characterize vegetation structure among areas with distinct fire histories and successional stages, we also extracted vegetation density values from the 2015 LiDAR data within both the reburn and reference areas (Figure 3).

We defined four seral stages based on the recent fire history of each sample point (Figure 5). We considered areas with no history of high-severity fire to be "late-seral" vegetation, comprised of two classes: "burned" forest, which experienced low/moderate severity fire, or "suppressed" forest if it had not burned since at least 1932. Conversely, areas that experienced high-severity wildfire were considered "early-seral" vegetation. Previous research in our study area suggested that these areas are typically composed of montane chaparral with scattered conifers or hardwoods present in low abundance (Cocking et al. 2014, Coppoletta et al. 2016). Early-seral vegetation created by the Storrie Fire, that was subsequently reburned at low/moderate severity in the Chips Fire, was characterized as potentially "recovering". This was based on the assumption that successional processes were not completely reset by low/moderate severity effects in the reburn; however, it is important to note that whether vegetation ultimately succeeds back to a mature forest is dependent on a number of unobserved factors, such as conifer tree seed availability, suitable site and climatic conditions, and future disturbance.

To characterize vegetation structure, we summarized vegetation density for each of the three strata within 125 m of sample points. We used a 125 m radius to match the scale of the highest performing late-seral and early-seral models described below. To test our assumption that initial high-severity fire transitions forests to markedly different vegetation structure, we compared once-burned early-seral points (n = 758) with once-burned late-seral points (n = 1413). To test how fire and succession reorganize vegetation structure within seral stages, we compared: (a) forests burned once at low/moderate severity (n = 1413) with unburned suppressed forest (n = 335); and (b) twice-burned early-seral points (i.e. two high-severity events; n = 119) with transitioning points that experienced 12 years of succession between an initial high-severity fire and a subsequent low/moderate-severity reburn (n = 70). Contrasts were made using linear models where the response variable was the mean or standard deviation of vegetation density within a 125 m radius of each point for each stratum, and seral stage was the sole categorical predictor.

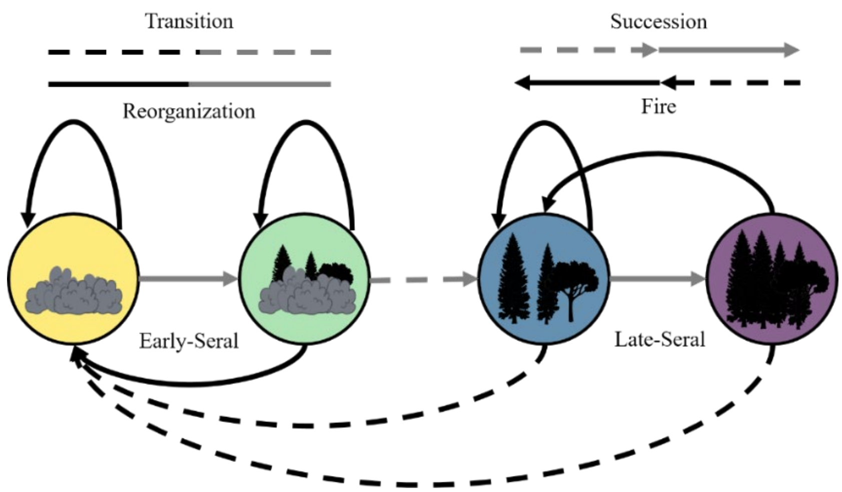

Figure 5. Illustration of the dynamic effects of fire and succession on vegetation seral stage.Fire and sustained succession can transition vegetation to an alternative seral stage (dashed lines) or can maintain the current state but reorganize its structure (solid lines) when conditions make vegetation resistant to type change. Given tree seed availability and suitable site and climatic conditions, early-seral vegetation reorganizes from shrub-dominated (yellow) to a mix of shrub species and young trees (green). Absent severe disturbance, succession will ultimately lead to late-seral forests. Historically, frequent low/moderate-severity fire maintained forests at relatively low tree densities and high spatial heterogeneity (blue), but through prolonged fire suppression many areas have grown uncharacteristically dense and homogenous (purple).

To assess the controls on fire-driven transition vs. stability we built two classes of linear models for points characterized as 1) early-seral vegetation (n = 747) and 2) late-seral vegetation (n = 1325) prior to the reburn. Specifically, points that burned at high severity (>75% canopy mortality) in the Storrie Fire were included in the early-seral models, while those that burned initially at low to moderate severity were included in the late-seral models. For each class of model and scale (150, 125, 100, 75, 50, or 25 m radius) we fit the following model to assess the effects of vegetation structure, topography and weather on the likelihood of a point reburning at high severity:

$$\begin{aligned} high\_severity_i \sim \quad & Bern(\phi_i) \\ logit(\phi_i) = \quad & \alpha_0 + \\ & \beta_{under.mn} * X_{1,i} + \beta_{under.sd} * X_{2,i} + \\ & \beta_{sub.mn} * X_{3,i} + \beta_{sub.sd} * X_{4,i} + \\ & \beta_{canopy.mn} * X_{5,i} + \beta_{canopy.sd} * X_{6,i} + \\ & \beta_{elev} * X_{7,i} + \beta_{rough} * X_{8,i} + \beta_{TWI} * X_{9,i} + \\ & \beta_{RH} * X_{10,i} + \beta_{wind} * X_{11,i} + \\ & GP(x_i, y_i) \end{aligned}$$

Where $high\_severity_i$ is a binary response variable indicating whether point *i* burned at high severity and α_0 represents the model intercept. Predictor variables describing vegetation structure included mean density ($\beta_{under.mn}$) and variation ($\beta_{under.sd}$) of understory vegetation, sub-canopy vegetation ($\beta_{sub.mn}$ and $\beta_{sub.sd}$), and canopy vegetation ($\beta_{canopy.mn}$ and $\beta_{canopy.sd}$). Variables describing topography include mean elevation (β_{elev}), topographic roughness (β_{rough}), and topographic wetness index (β_{TWI}). Fire weather variables included relative humidity (β_{RH}) and wind speed (β_{wind}). We applied a Bernoulli error structure with a logit link and a partial gaussian process estimating the spatial covariance among points where x_i and y_i represent sample point coordinates.

We compared predictive performance of models of different spatial scales using the leave-one-out information criteria (Vehtari et al. 2017). Models were fit using the brms and rstan packages (Bürkner 2017, Stan Development Team 2018) in the R statistical environment (R Core Team 2020). The full joint model was run with 4 chains, each for 2000 samples with a warmup of 1000 samples and 4000 total post-warmup samples. Traceplots and R-hat values were assessed for proper mixing and model convergence. Bayesian contrasts were used to calculate the probability that the two distributions were different.

Aerial imagery

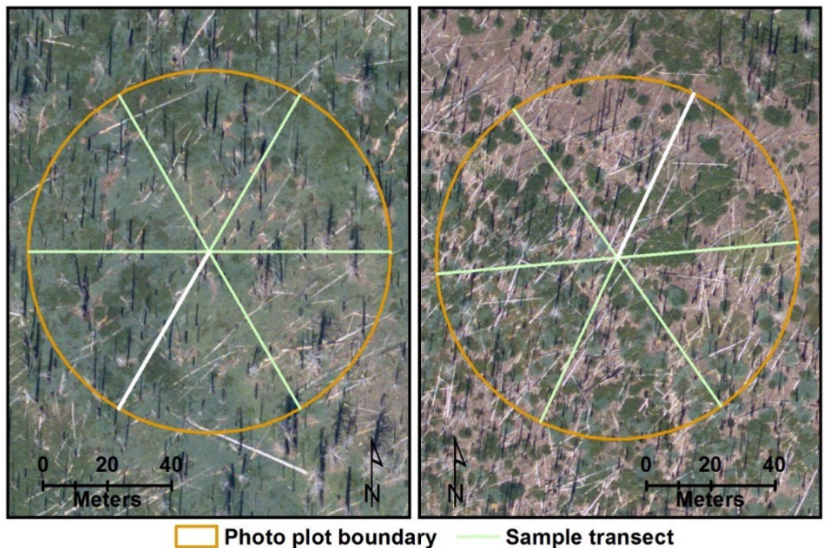

Figure 6. Examples of two 1-ha orthoimagery plots(scale: 1:1200)

Since we were unable to directly evaluate the effect of post-fire management (Objective 2) due to inconsistencies among treatments, we refocused our analysis to evaluate those attributes of the post-fire landscape that managers can directly influence: snags, large woody fuels, shrubs, and trees. We used high resolution (0.3 m) aerial orthoimagery, collected within the footprint of the Storrie Fire in August 2009 (Quantum Spatial, Novato, California). We randomly selected 202 circular 1 ha plots in ArcMap 10.3.1 (ESRI 2014) within high-severity patches (RdNBR ≥ 641), with a minimum spacing of 200 m between plot centers. We limited our analysis to high-severity patches in the Storrie Fire because the loss of overstory cover allowed for more accurate visual assessment of understory fuels conditions, including snags, logs, and shrub cover, as compared to areas with intact tree canopy. We restricted the analysis to US Forest Service land, included all vegetation types, and excluded areas burned by wildfire between 2000-2009.

Snag density, log density, shrub cover, and live tree cover were assessed visually within the 1 ha orthoimagery plots by two photo analysts. All visible snags were counted for the entire plot. Visible logs were tallied by intersection along six 56.4 m radial transects (Figure 6). To provide an estimate of the total number of logs per plot, we used simple linear regression in a subset (n=41) of plots to derive the relationship between the number of logs tallied on the transects and a count of the total number of logs in the plot. Cover of shrubs and live trees within each plot were estimated visually within six subsections created by the plot transects, then averaged for the plot. Plots were assigned to analysts randomly so that each analyst provided data for plots across the entire Storrie Fire, covering a range of elevation, slope, and aspect. Twenty plots were measured by both photo analysts to allow for a comparison of estimates between operators.

Imagery Analysis

We assessed variation in three response variables: snag density, log density, and shrub cover. We first used generalized additive models (GAMs) to assess how post-fire fuels varied with topographic and water balance variables across the Storrie Fire footprint (Table 4). Second, we used ANOVA to detect differences in fuels between fire severity classes in the subsequent 2012 Chips Fire. We also compared fuel conditions in plots that burned at high severity in the Storrie Fire and were unburned by the Chips Fire. For more detail on the methodology, refer to Lydersen et al. 2019.

Table 4. Covariates used in GAMs to assess variation in post-fire fuels nine years after the Storrie Fire.

Model covariate	Mean (min-max), or class (N)
Elevation	1440 (838–1989)
Topographic position index	Canyon (21), flat slope (6), steep slope (117), ridge (58)
Slope steepness (%)	51.1 (4.7–136.5)
Solar radiation (WHm-2 x 105)	13.1 (5.7–16.8)
Slope aspect (transformed)	1.03 (0–2)
Actual evapotranspiration	385 (293–534)
Climatic water deficit	582 (359–703)

Results and Discussion

Field plots

Descriptive Analysis

We found significant differences ($p < 0.05$) between pre- and post-reburn samples for four of the five response variables (Table 5). Across all plots, density of mature trees and shade tolerance index (STI) decreased significantly after the Chips Fire reburn, while seedling density and shrub cover increased. There was no significant change in fire tolerance index (FTI) between pre- and post-reburn samples. In plots reburned by the Chips Fire that had trees remaining (trees $ha^{-1} > 0$), tree density significantly decreased (270.45 to 169.20 trees ha^{-1}, $p < 0.05$), but neither FTI nor STI were significantly different between pre- and post-Chips Fire samples.

Table 5. Means ± S.E. across all sample plots included in the GLM analyses; p-value based on Wilcoxon Signed-Rank Test of significant differences ($\alpha = 0.05$) between paired pre- and post-Chips Fire samples.

Dependent variables	Pre-reburn	Post-reburn	p-value
Trees ha^{-1}	188.34±18.04	121.58±13.33	<0.001
FTI	400.46±24.69	346.37±26.39	0.304
STI	878.22±49.35	705.37±51.67	<0.001
Seedlings ha^{-1}	4356.35±1290.55	7552.95±1580.98	<0.001
Shrub cover (%)	25.46±2.48	29.92±2.38	0.004

Species-wise comparisons of importance values between pre- and post-reburn samples showed only minimal variation between sample periods when plots reburned at low severity or were unchanged Figure 7). Moderate severity reburn had a more substantial effect, resulting in greater mortality across all initial severity combinations (Figure 7); however, the effect of moderate severity reburn on species composition was inconsistent. Fir importance increased and pine was extirpated in the unchanged/moderate interaction. Fir and pine importance both decreased, while Douglas fir importance remained the same, in the low/moderate severity interaction. Fir, incense cedar, and Douglas fir importance decreased, while pine exhibited little change, in the moderate/moderate interaction.

Across all plots (n=134) and fire severities, mean (4356 ha^{-1}) and median (347 ha^{-1}) seedling densities prior to the reburn were less than the mean (7553 ha^{-1}) and median (1442 ha^{-1}) after the reburn. Approximately 85% of plots supported conifer seedlings after the Chips Fire, compared to 68% of plots prior to the reburn. The greatest percent frequency of seedlings was found in plots that were unchanged by the first fire and burned at low severity in the Chips Fire. When we limited our analysis to include only those plots where seedlings occurred, we found higher mean and median seedling densities after the reburn, compared to prior to the reburn, for pine, true fir, incense cedar, and all mixed conifer species combined (Figure 8). In contrast, the density of Douglas fir seedlings in plots was higher prior to the reburn than after. True fir seedling densities were higher (both pre- and post-reburn) than both pine and incense cedar. Following the reburn, some seedlings were recorded in plots with no standing mature trees up to a distance of 188 m from the closest potential seed source, and seedling density per sampling year closely tracked yearly precipitation (Figure 9).

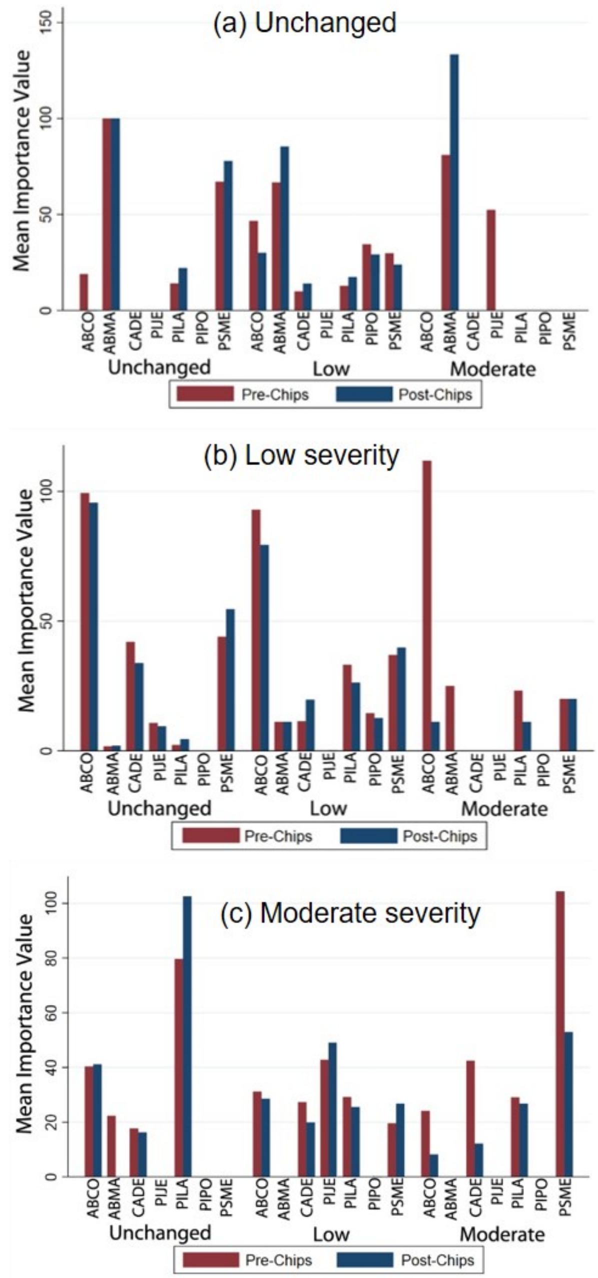

Figure 7. Species-wise variation in mean importance value for all conifer species in plots. Graphs are divided based on initial severity category: a) unchanged, b) low severity, and c) moderate severity; values on the x-axis are grouped by reburn (Chips Fire) severity category. Species abbreviations found in Table 2.

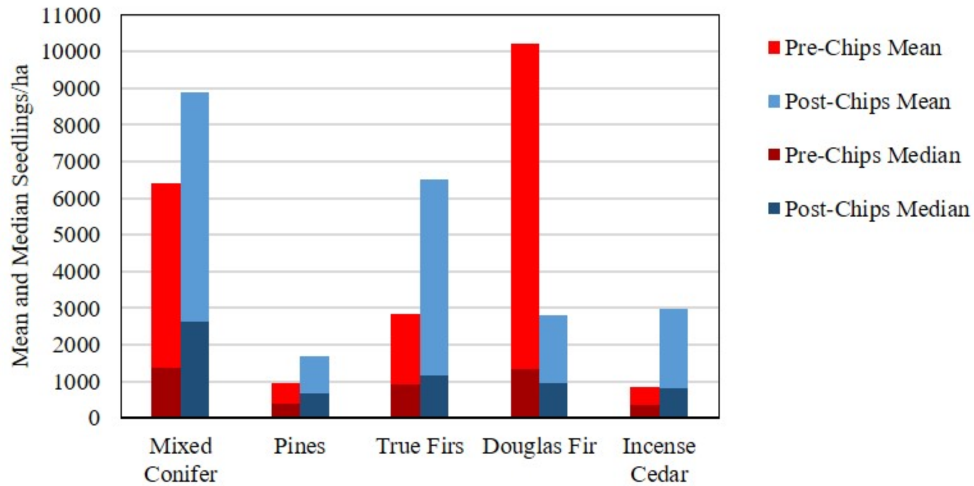

Figure 8. Species composition patterns in conifer regeneration pre- and post-reburn. Mean and median seedlings ha^{-1}, calculated using only those plots where seedlings occurred, for all mixed conifers and each genus in pre-Chips and post-Chips Fire samples.

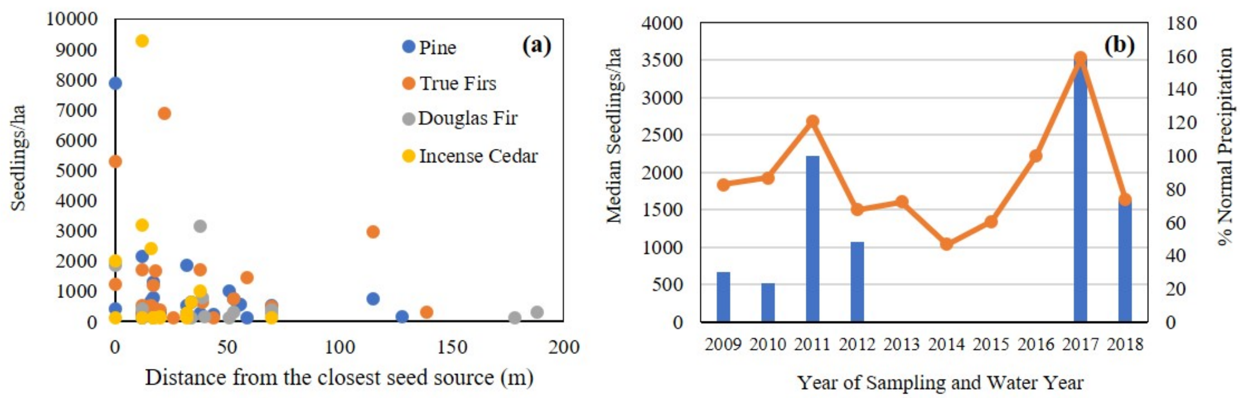

Figure 9. Graphs describing patterns of seed dispersal and annual regeneration within the study area. (a) Post-Chips seedlings ha^{-1} by genus in high-severity plots, without a live seed source within the plots, plotted against distance from the closest mature live mixed conifer seed source (plots with density >10,000 seedlings ha^{-1} excluded, n = 3). (b) Median seedlings ha^{-1} by year of sampling only for plots where seedlings occurred (blue bars), and percent normal (1980-2010) precipitation for each water year (orange line; weather station used was Bucks Creek).

Tree density

Changes in tree density were significantly related to the interactions between initial and reburn severity, as well as shrub cover prior to the reburn ($R^2 = 0.57$, $p < 0.05$). Figure 10 illustrates the predicted change in tree density for each fire severity combination. Significant decreases in canopy tree density were predicted in high-severity reburn plots that were initially unburned (average change: -625 trees ha^{-1}), unchanged (average change: -608 trees ha^{-1}), or burned at low severity in the initial fires (average change: -238 trees ha^{-1}). Significantly reduced tree densities were predicted in moderate severity reburn plots across all initial severity categories (average change: -184 to -128 trees ha^{-1}), with the exception of high initial fire severity. No significant changes in tree density were predicted in plots initially burned at moderate and high severities and reburned at high severity because density was already substantially reduced by the initial fires. Significantly reduced tree density was predicted in low-severity reburn plots burned at low (mean -70 trees ha^{-1}) and high (mean -163 trees ha^{-1}) initial severities.

In our study, changes in tree density were significantly related to pre-Chips Fire shrub cover. In dry mixed conifer forests, understory shrub cover is generally lower in dense forested stands with high canopy closure. Plots with these conditions were also more likely to have substantial change in tree density than plots that had lower tree densities and higher cover of shrubs prior to the reburn.

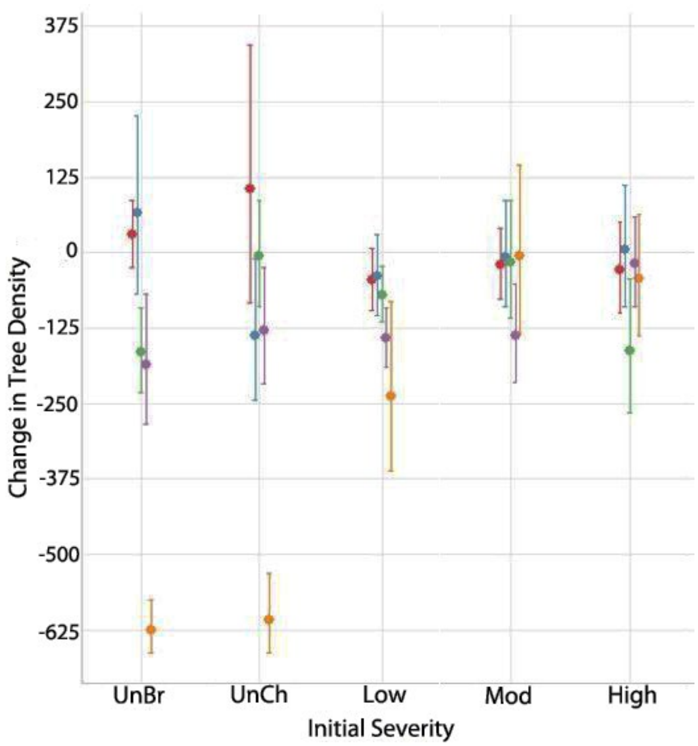

Figure 10. Effect plot of change from pre- to post-reburn samples in mixed conifer live tree density (trees ha^{-1}) for all severity combinations. Chips Fire reburn severity symbology (initial severity abbreviations in parenthesis): unburned (UnBr) = red, unchanged (UnCh) = blue, low = green, moderate (Mod) = purple, high = orange, and the error bars represent 95% confidence intervals. Response variable was back transformed to the original units, thus zero on the Y-axis indicates no change from pre- to post-Chips Fire samples.

Fire and shade tolerance

Based on the reductions in tree density, we expected tree species composition to shift from greater proportions of shade tolerant/fire intolerant to more shade intolerant/fire tolerant species in plots reburned at low to moderate severity. However, plots that supported live overstory trees after the reburn showed very little change (from pre-reburn levels) in fire or shade tolerance indices. Reductions in fire tolerance and shade tolerance did occur following the reburn; however, these changes were largely driven by almost complete mortality of most overstory trees due to moderate- and high-severity reburn (Figure 11).

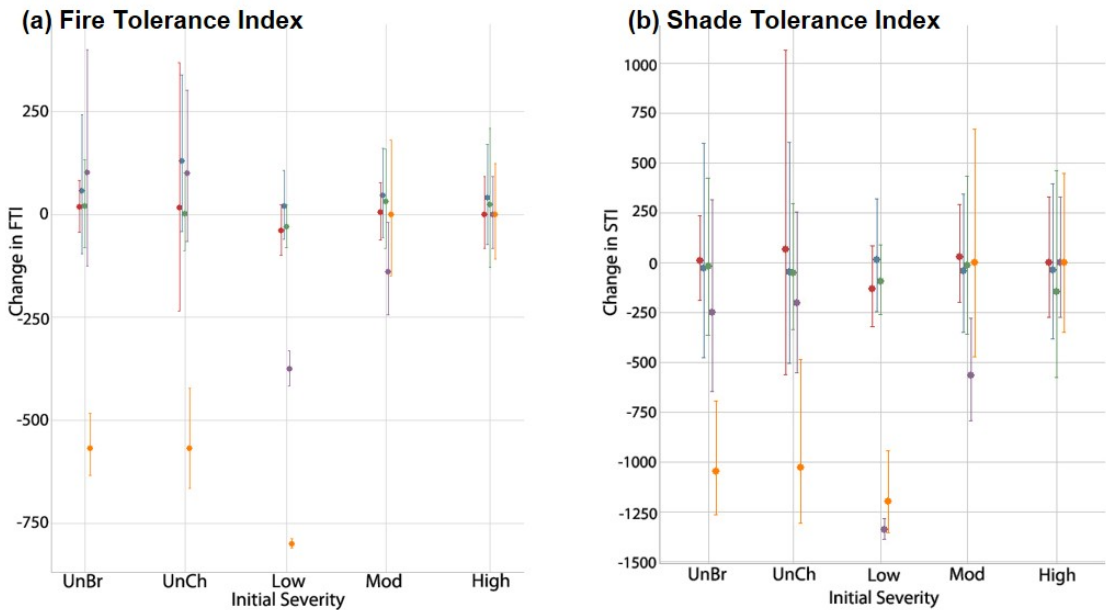

Figure 11. Effect plots of change from pre- to post-reburn samples in (a) Fire Tolerance Index and (b) Shade Tolerance Index for all severity interactions. Chips Fire reburn severity symbology (initial severity abbreviations in parenthesis): unburned (UnBr) = red, unchanged (UnCh) = blue, low = green, moderate (Mod) = purple, high = orange, and the error bars represent 95% confidence intervals.

Seedling density

There were no significant pre- or post-Chips Fire severity categories or interactions ($\alpha < 0.05$) in the GLM (although the interaction of low initial and low reburn severities had $p = 0.07$). However, the margins plot 95% confidence interval for the low/low interaction did not overlap zero, predicting a positive change in seedling density (mean +6,678) (Figure 12a).

Shrub cover

Significant reductions in shrub cover (average change: -19% cover) were predicted in plots repeatedly burned at high severity, while significant increases were predicted at various lower-severity combinations (Figure 12b). In areas outside of the reburn (i.e. once-burned plots), significant shrub cover increases were predicted following a moderate severity burn (mean change: +16% cover). A significant increase was also predicted in plots that initially burned at high severity and were classified as unchanged by the reburn (average change: +22% cover) and plots that burned at low severity in the initial fire and moderate severity in the reburn (average change: +17% cover).

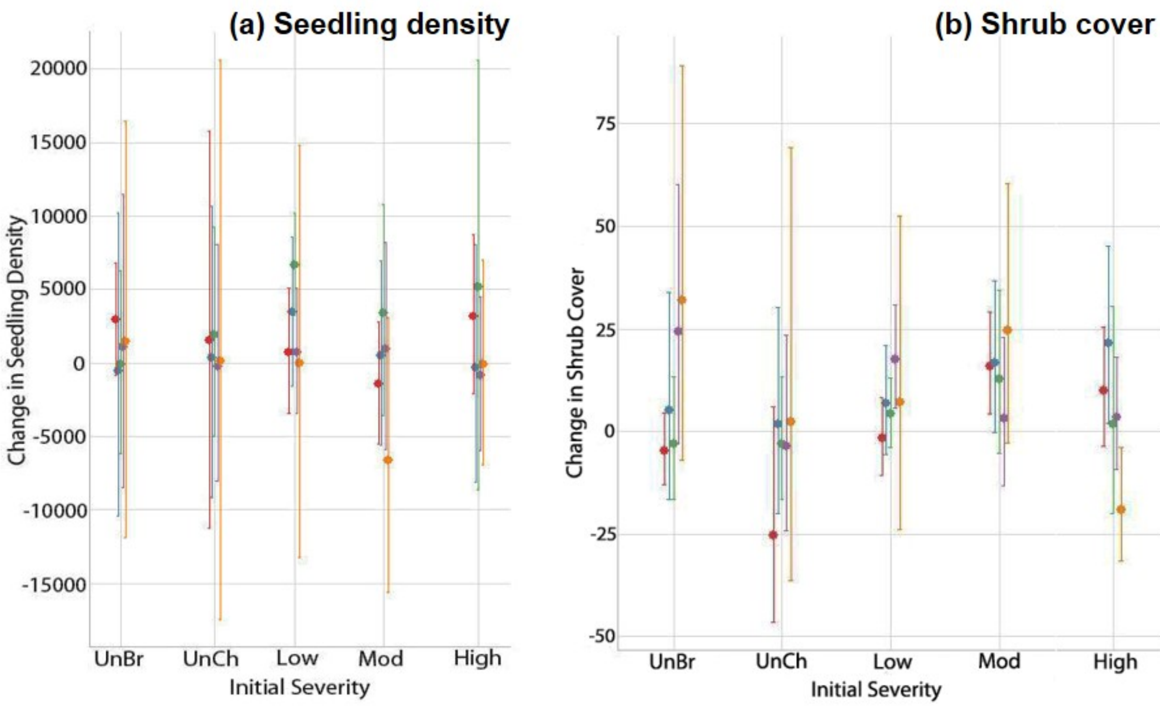

Figure 12. Effect plots of change from pre- to post-Chips Fire samples in (a) mixed conifer seedlings ha^{-1} and (b) shrub cover ha^{-1} for all severity interactions. Chips reburn severity symbology (initial severity abbreviations in parenthesis): unburned (UnBr) = red, unchanged (UnCh) = blue, low = green, moderate (Mod) = purple, high = orange, and the error bars represent 95% confidence intervals.

LiDAR

Within the area of overlap between the Storrie and Chips fires, approximately 35% of fire-excluded mixed conifer forest was converted to early-seral vegetation by high-severity fire effects in the initial fire (Figure 13). Approximately 66% of these early-seral vegetation types reburned at high severity during the Chips Fire. The remaining 34% of early-seral vegetation types (i.e. areas initially burned at high severity) reburned at lower severities in the Chips Fire. An estimated 65% of fire-excluded mixed conifer forest in our study area burned at low to moderate severity in the initial Storrie Fire; approximately 92% of these areas reburned again at low to moderate severity during the Chips Fire (Figure 13).

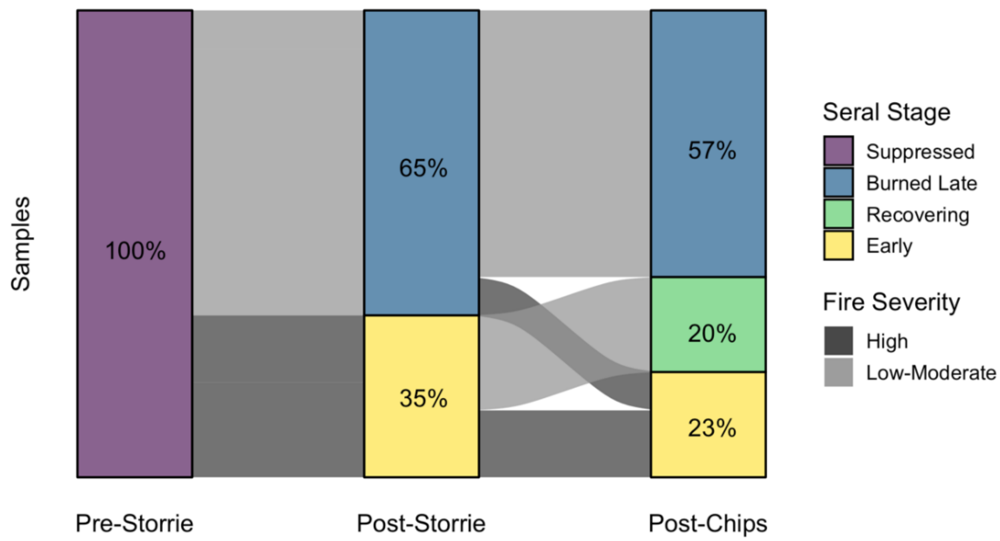

Figure 13. Stability, reorganization, and transition of vegetation within the area of overlap between the 2000 Storrie Fire and 2012 Chips Fire reburn. Sample points were limited to areas of lower montane forest prior to the initial fire.

<u>Vegetation Structure</u>

We found clear structural differences between the early-seral and late-seral groups following a single fire. Not surprisingly, areas characterized as late-seral forest had higher vegetation density in the canopy and lower density in the understory compared to the early-seral group. Spatial complexity in both the canopy and understory was greater for the late-seral type compared to the early-seral group. We found only small differences in density and heterogeneity of the sub-canopy between the two seral groups (Figure 14).

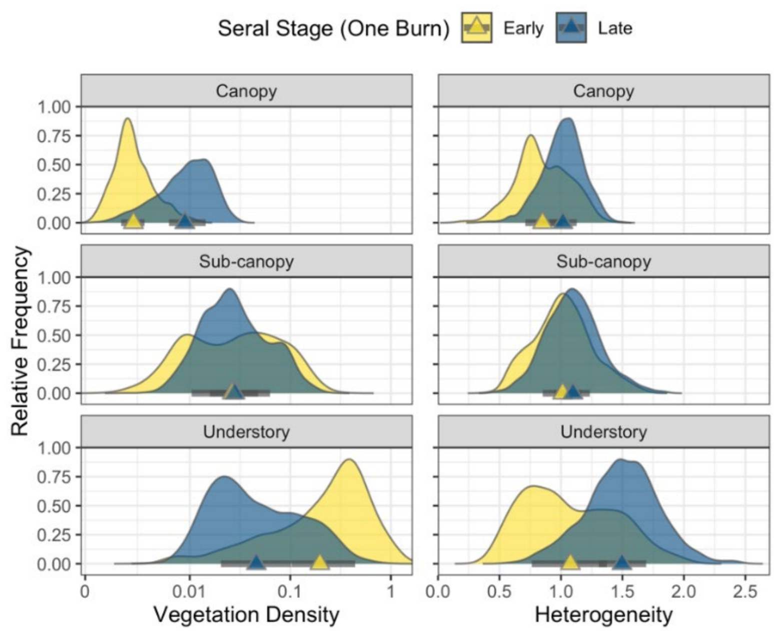

Figure 14. Sample distribution plots for once burned early- and late-seral vegetation as measured within three vertical strata. Triangles and bars below distributions indicate mean and 50% inter-quantile ranges, respectively. The horizontal axis indicates vegetation density on the log scale (first column) and the standard deviation of within sample density (second column). The vertical axis indicates the frequency of samples relative to a panel's distribution maximum.

We also observed clear structural differences between once-burned and long unburned (i.e. suppressed) late-seral points. Unburned forest, with no evidence of recent fire, had higher density of vegetation in the canopy and sub-canopy strata than once-burned late-seral points. In contrast, the density of understory vegetation was slightly higher in the burned late-seral points. Across all three vertical strata, once-burned late-seral vegetation had higher within-sample heterogeneity than fire suppressed forests (Figure 15).

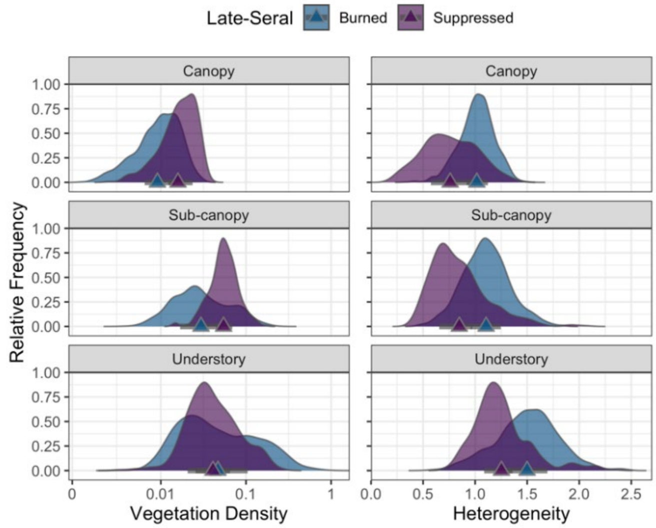

Figure 15. Sample distribution plots for once burned late-seral and fire suppressed (unburned) vegetation as measured within three vertical strata. Triangles and bars below distributions indicate mean and 50% inter-quantile ranges, respectively. The horizontal axis indicates vegetation density on the log scale (first column) and the standard deviation of within sample density (second column). The vertical axis indicates the frequency of samples relative to a panel's distribution maximum.

Following the second wildfire, points classified as potentially recovering (i.e. those that burned initially at high severity then reburned at low/moderate severity) had higher vegetation density and heterogeneity in both the canopy and sub-canopy as compared to early-seral points that experienced repeated high-severity fire; however the absolute differences were small due to the sparse upper strata of the two classes (Figure 16). Differences in density and heterogeneity between the two seral groups were also small for the understory strata (Figure 16).

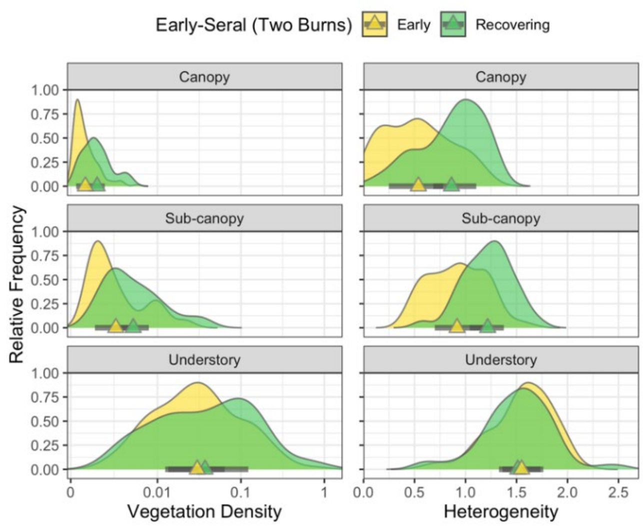

Figure 16. Sample distribution plots for twice burned early-seral (both at high severity) and potentially recovering (high then low/moderate severity) vegetation as measured within three vertical strata. Triangles and bars below distributions indicate mean and 50% inter-quantile ranges, respectively. The horizontal axis indicates vegetation density on the log scale (first column) and the standard deviation of within sample density (second column). The vertical axis indicates the frequency of samples relative to a panel's distribution maximum.

Controls on resilience

The final early-seral model (circular radii: 125 m) showed that vegetation structure and topography, but not fire weather, were important predictors of reburn severity. The likelihood of reburning at high severity was estimated to decline (Pr. ≥ 99%) with increasing topographic roughness, sub-canopy density and heterogeneity, and topographic wetness index (Figure 17). In contrast, understory density was positively associated with high severity reburns (Figure 17). Neither wind speed, relative humidity, elevation, canopy density, nor canopy heterogeneity showed clear effects (Pr. < 82%). Taken together, these results suggest that areas initially burned at high severity, which were situated in variable and mesic terrain, and characterized by relatively sparse understory vegetation and heterogeneous sub-canopy vegetation, were most likely to avoid repeat high-severity fire and continue recovery toward late-seral forest conditions.

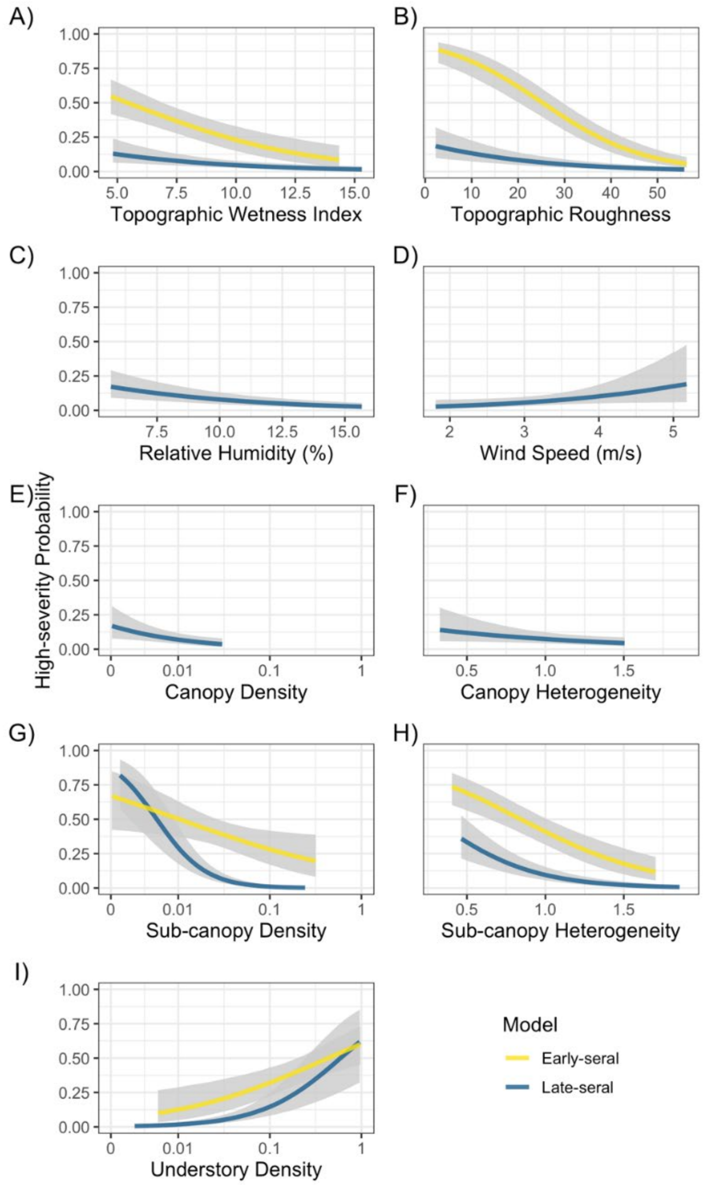

Figure 17. Marginal effects plots for early-seral (yellow) and late-seral (blue) models. Density values in E, G, and I represent mean vegetation area density. Heterogeneity values in F and H represent the standard deviation of density. Lines represent mean estimates and bands show 90% credible intervals. Only predictor variables with at least a 95% probability of a positive or negative effect are shown.

The final late-seral model (circular radii: 125 m) indicated that vegetation structure, fire weather, and topography all influenced the likelihood of burning at high severity during the second fire. The likelihood of once-burned late-seral vegetation reburning at high severity was estimated to decline (Pr. ≥ 96%) with increased sub-canopy density and heterogeneity, topographic roughness, relative humidity, topographic wetness index, and canopy density and heterogeneity (Figure 17). Conversely, understory density was positively associated with high severity in the second burn (Pr. > 99%; Figure 17). Neither understory heterogeneity, nor elevation showed clear effects (Pr. < 90%). Taken together, these results indicate that once-burned forests characterized by relatively dense but heterogeneous upper strata and sparse understory, located in variable and mesic terrain, and burning under moderate fire weather conditions, are most likely to resist high severity reburns.

Our finding that vegetation density in the subcanopy of late-seral vegetation types was negatively associated with high severity reburn was surprising, particularly because understory and subcanopy vegetation are often characterized as ladder fuels, which are generally associated with increased risk of high-severity fire in fire-excluded forests. Plausible mechanisms for this dynamic include: differential selection for more fire-resistant tree species with elevated crown base heights following the initial low/moderate severity fire; suppression of understory growth due to the moderately dense overstory; reduction in surface-level airflow due the presence of sub-canopy vegetation; and promotion of higher fuel moisture at the forest floor due to shading from the overstory and sub-canopy (Bigelow and North 2011, Estes et al. 2012, Ziegler et al. 2020).

Aerial imagery

Among the topographic and water balance variables considered in the GAMs, only elevation and slope steepness had a significant effect on fuel variability. Snags and logs both tended to be more numerous on gentler slopes, which may have supported greater pre-fire tree densities due to deeper soils with greater water holding potential (Meyer et al. 2007). Greater snag density also occurred at mid-level elevation (approximately 1400 m). We found that shrub cover decreased with increasing elevation, although the GAM only accounted for 6% of the variation in shrub cover. These results are consistent with the effect of elevation on productivity, where lower elevation sites are often too dry to support greater forest cover and higher elevation sites may be more limited due to colder temperatures. The lower elevations at our study site also tend to occur in steep canyons where soil depth may limit productivity.

We found that high densities of both snags and logs were associated with high severity in the subsequent fire (Figure 18). The density of both snags and logs was significantly higher in plots that reburned at high severity compared to low and moderate severity, and areas outside the Chips Fire (Figure 18). Shrub cover had a marginally insignificant (P = 0.0515) effect on subsequent fire severity, with generally higher shrub cover associated with higher-severity fire. This marginal effect may have been due to variability among analysts; in our comparison among analysts, shrub cover was greater in plots that reburned at high severity compared to low severity for one analyst.

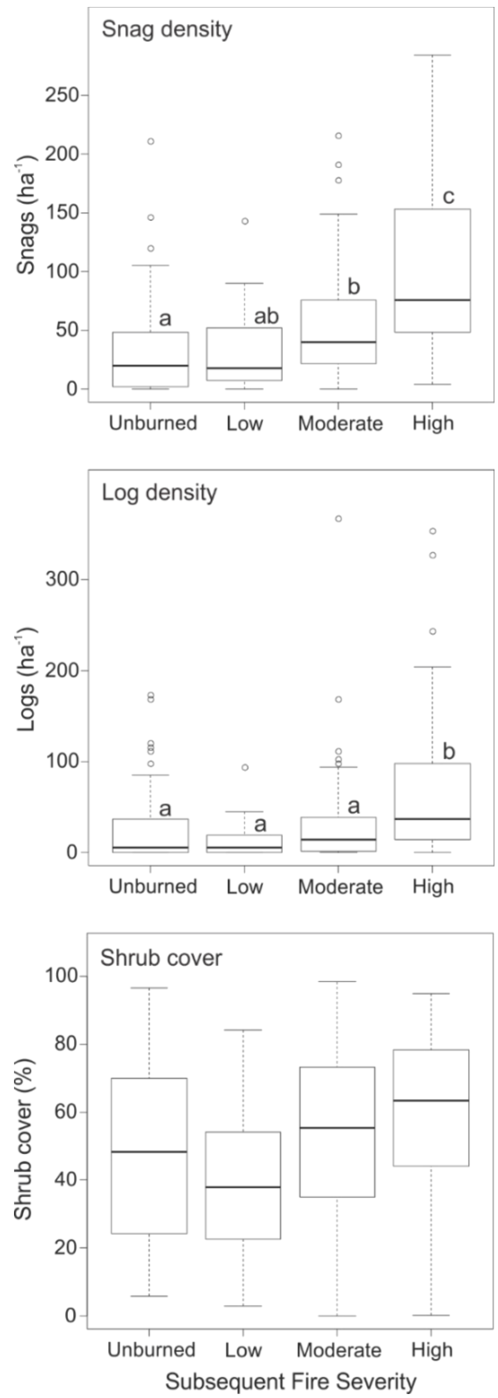

Figure 18. Fuel loads in 2009 by reburn severity class, for areas initially burned at high severity in the 2000 Storrie Fire. Unburned plots were outside of the Chips Fire. Different letters indicate significant differences between subsequent fire severity classes. Box and whisker plots depict median (horizontal band), interquartile range (white bar), range of data within 1.5 interquartile range of the lower and upper quartiles (vertical dashed lines), and outliers (open circles).

Science delivery, products, future planned actions

One of the most substantial impacts of the initial reduction in funding was on our ability to host a large workshop for managers. Nevertheless, we worked closely with managers and other scientists, to discuss and disseminate relevant findings from this study. Scientific delivery products include eight oral presentations and three contributed poster presentations at regional and national scientific meetings; we also have one published peer-reviewed journal article, one currently in review, and one additional journal article that is currently in preparation.

Figure 19. UC Berkeley students collecting field data as part of a field day and lecture designed to examine the effects of management and successive wildfires on forest vegetation and fuels.

Working closely with colleagues from UC Berkeley, we developed a full-day lecture and field exercise for forestry students that focused on examining areas burned by two successive wildfires and testing the hypotheses related to Objectives 1 and 2. We led this lecture and field exercise in 2017, 2018, and 2019, and plan to continue this effort into the future.

To assist with outreach and education, we developed a project website, hosted by the Department of Geosciences at Florida Atlantic University (http://www.geosciences.fau.edu/reburn/). This website features photos and brief descriptions of the project objectives, methodologies, and outcomes. This site is regularly updated and contains links to presentations, publications, and other deliverables as they become available.

Throughout the project, we worked closely with resource managers and interdisciplinary teams on the Plumas and Lassen national forests to incorporate relevant project findings into actual on-the-ground post-fire restoration and management. One outcome of this was the *Clustered Lady's Slipper and Serpentine Rare Plant Community Conservation Project*, which is currently being implemented within the Storrie and Chips Fire perimeters to reduce the potential for high severity reburn in sensitive rare plant habitats. Additionally, the findings from this project are being incorporated into current post-fire restoration plans (e.g., 2020 Walker Fire Recovery Project).

Future Actions

Funding from this JFSP grant enabled the collection of a large quantity of field data, only a fraction of which were presented in the analyses described in this report. We plan to continue (and expand) our efforts to examine the effects of successive fires on hardwoods, changes in shrub species composition, and fuel rearrangement. We are also working with Lassen and Plumas National Forest staff to identify accurate and representative post-fire treatments, situated outside of these fire footprints, that will enable us to conduct a direct assessment of post-fire management on subsequent fire severity and ecological outcomes.

Conclusions

Reburning on relatively short intervals (< 15 years) is becoming increasingly common in mixed conifer forests across the western U.S. This study took advantage of an early opportunity to study this phenomenon, and in doing so identified several management-relevant pathways by which post-fire vegetation structure and fuels influence the severity and ecological outcomes of a subsequent wildfire.

Following the initial fire, we found clear structural differences between areas burned at high severity and areas that were unburned or burned at lesser severities. Areas that initially burned at high severity were characteristic of early-seral vegetation types, with very few live trees in the overstory, dense shrub vegetation in the understory, large woody fuels, and low conifer regeneration (i.e. below recommended stocking densities; USDA Forest Service 1989). In contrast, areas that burned initially at low to moderate severity had characteristics that more closely aligned with resilient late-seral forest conditions, such as lower tree densities, higher spatial complexity in canopy and understory vegetation and variable, but generally higher, conifer regeneration. Compared to adjacent fire-excluded stands, forests that experienced a single low to moderate severity fire had less dense canopy and subcanopy vegetation and higher variability in all height strata.

Surprisingly, both early-seral and late-seral vegetation types demonstrated an ability to resist high-severity fire when reburned within a short timeframe (i.e. 12 years). This resistance was contingent on a combination of factors, including topography, fire weather during the reburn, vegetation structure, and woody fuels. In our study, topography played an important role in moderating reburn severity, with more variable and mesic terrain having a lower probability of high severity reburn. In forests that were burned at low to moderate severity in the initial fire, high relative humidity and low wind speeds during the reburn also reduced the likelihood of high-severity fire effects.

While factors like fire weather and topography are undoubtedly important drivers of fire severity patterns, our analysis of high-resolution LiDAR data and aerial imagery demonstrated that woody fuels and vegetation structure can also influence fire severity in a subsequent reburn. In areas initially burned at high severity, we found that high densities of both snags and down woody fuels were associated with high severity effects in the second fire. Shrub cover was also generally associated with higher severity reburn.

Variability in vegetation structure played an important role in moderating severity in the second fire. In early-seral habitats, areas with relatively sparse understory and variable sub-canopy were most likely to avoid repeat high-severity fire and continue recovery toward a forested condition. Late-seral forests that burned at low to moderate severity in the initial fire were most likely to resist high-severity fire if they were characterized by sparse understories (< 2 m in height) and relatively dense but heterogeneous vegetation in the upper strata (> 2 m in height).

The results of our analysis of pre- and post-reburn field plots and high-resolution LiDAR data generally align with the ecological pathways proposed in our conceptual model of post-fire vegetation and fuel dynamics following initial fires and reburns (Figure 1, Objective 1). Within the area of overlap between the Storrie and Chips fires, approximately 35% of fire-excluded mixed conifer forest was converted to early seral vegetation by high-severity fire effects in the initial fire; approximately 66% of these early-seral vegetation types reburned at high severity during the Chips Fire. Our plot-level analysis indicated that areas impacted by successive high-severity fires had little to no live conifer overstory, retained high cover of shrubs in the understory, and had little to no conifer regeneration. The pattern of repeat high-severity fire and observed ecological outcomes, provide evidence of a positive feedback loop, one that has the potential for long-term state change from confer forest to persistent montane chaparral.

Figure 20. Example of early-seral habitat created and maintained by repeat high-severity fire.

Approximately 34% of the areas that initially burned at high severity, reburned at lower severities in the Chips Fire. Our LiDAR analyses revealed significant structural differences between these areas (high severity followed by low/moderate severity) and persistent early-seral vegetation types (two high-severity events), with the former characterized by higher density and heterogeneity in sub-canopy and canopy vegetation. These results suggest that early-seral habitats that burn at lower severity in a second wildfire may be in the early stages of forest recovery. We found limited evidence for this in the field portion of this JFSP study, which found that recovering areas supported a diverse mixture of shrub species in the understory and re-sprouting black oak and conifers in the subcanopy. However, it is important to note that whether succession will ultimately lead to recovery of late-seral forests, will largely depend on conifer tree seed availability and suitable site and climatic conditions. In the near-term, many of the areas impacted by large patches of high-severity fire are likely be dominated by re-sprouting hardwood species (e.g. black oak) in the emerging sub-canopy and canopy strata.

We found evidence of stabilizing negative feedback loops in areas that burned at low to moderate severity in both the initial fire and reburn. An estimated 65% of fire-excluded mixed conifer forest in our study area burned at low to moderate severity in the initial Storrie Fire; approximately 92% of these areas reburned again at low to moderate severity during the Chips Fire. Our analysis of field data demonstrated that these successive low to moderate severity fires significantly reduced tree density, increased tree regeneration, and in some severity combinations (i.e. low followed by moderate severity) promoted colonization by shrubs. We did not find evidence that multiple low to moderate fires were able to shift forest composition toward dominance of more fire tolerant or shade intolerant species.

Management implications

We were not able to directly assess the effects of post-fire management actions on subsequent fire severity patterns; however, we were able to analyze the influence of vegetation and fuels, across a wide range of post-fire conditions, on fire severity patterns and ecological outcomes. Taken together, the results of our analyses demonstrate that vegetation structure and fuel loads, which are often targeted in post-fire restoration projects, are also important drivers of subsequent fire severity and post-reburn vegetation structure.

Figure 21. Regenerating pine in an area burned twice at low to moderate severity.

In areas where high severity reburn is undesirable, managers can utilize treatments like thinning or mastication to break up vertical and horizontal continuity of vegetation; salvage logging to reduce the density of snags; and machine piling or prescribed fire to reduce the quantity or influence the arrangement of large woody surface fuels. In some areas, proactive reforestation may be necessary to facilitate the transition from early-seral vegetation toward a later-seral forested condition. Our findings suggest that planting efforts may be most successful when focused on areas that are less likely to burn again at high severity. For example, planting in rough and mesic terrain (Figure 17), in the middle of large high-severity patches where conifer seed rain is limited (Shive et al. 2018), may be the most strategic use of resources (North et al. 2019).

Although the majority of post-fire restoration efforts are focused on areas that burn at high severity, our study suggests that management activities in areas that have burned at low to moderate severity, may also be necessary to maintain or promote the restorative benefits of the first fire. We found evidence that multiple low to moderate severity fires in short succession (i.e. < 12 years) can restore forest structure by reducing tree densities and increasing vertical heterogeneity. However, we also found that repeat low to moderate severity fire may not be enough to shift species composition patterns. In some cases, management actions such as underburning or thinning, in areas that were unchanged or that burned at low to moderate severity will be needed to reduce the density of fire sensitive species (e.g. firs) and increase dominance of fire tolerant species (e.g. pines).

Literature Cited

Abatzoglou, J.T. 2013. Development of gridded surface meteorological data for ecological applications and modelling. International Journal of Climatology 33:121–131.

Albini, F.A. 1976. Estimating wildfire behavior and effects. Gen. Tech. Rep. INT-30. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station, Ogden, Utah.

Bigelow, S.W. and M.P. North. 2011. Effects of fuels-reduction and group-selection silviculture on understory microclimate in Sierran mixed-conifer forest. Forest Ecology and Management 264: 51-59.

Bouvier, M., S. Durrieu, R.A. Fournier, and J.P. Renaud. 2015. Generalizing predictive models of forest inventory attributes using an area-based approach with airborne LiDAR data. Remote Sensing of Environment 156:322–334.

Brown, J.K. 1974. Handbook for inventorying downed woody material. USDA Forest Service General Technical Report INT-16. U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station, Ogden, UT. 24 p.

Bürkner, P.C. 2017. brms: An R Package for Bayesian multilevel models using Stan. Journal of Statistical Software 80:1–28.

California Department of Forestry and Fire Protection. 2020. Fire Perimeters - California, California Department of Forestry & Fire Protection, Fire and Resource Assessment Program. http://frap.fire.ca.gov/data/frapgisdata-sw-fireperimeters_download.

CALVEG. 2004. [ESRI personal geodatabase]. USDA-Forest Service, Pacific Southwest Region, McClellan, CA.

Cocking, M.I., J.M. Varner, and E.E. Knapp. 2014. Long-term effects of fire severity on oak–conifer dynamics in the southern Cascades. Ecological Applications 24:94–107.

Collins, B.M., and C. Skinner. 2014. Fire and fuels. *In* J.W. Long, L. Quinn-Davidson, and C.N. Skinner, Eds. Science synthesis to support socioecological resilience in the Sierra Nevada and southern Cascade Range. U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, CA. pp: 143-172

Collins, B.M., and G.B. Roller. 2013. Early forest dynamics in stand-replacing fire patches in the northern Sierra Nevada, California, USA. Landscape Ecology 28:1801-1813.

Collins, B.M., M. Kelly, J.W. van Wagtendonk, and S.L. Stephens. 2007. Spatial patterns of large natural fires in Sierra Nevada wilderness areas. Landscape Ecology, 22 (4): 545–557.

Collins, B., J. Miller, A. Thode, M. Kelly, J. van Wagtendonk, and S. Stephens. 2009. Interactions among wildland fires in a long-established Sierra Nevada natural fire area. Ecosystems 12:114-128.

Coppoletta, M., K.E. Merriam, and B.M. Collins. 2016. Post-fire vegetation and fuel development influences fire severity patterns in reburns. Ecological Applications 26: 686-699.

Curtis, J.T. and R. McIntosh. 1951. An upland forest continuum in the prairie-forest border region of Wisconsin. Ecology 32: 476–496.

ESRI [Environmental Systems Research Institute]. 2014. ArcGIS release 10.3. Redlands: Environmental Systems Research Institute.

Estes, B.L., E.E. Knapp, C.N. Skinner, J.D. Miller, and H.K. Preisler. 2017. Factors influencing fire severity under moderate burning conditions in the Klamath Mountains, northern California, USA. Ecosphere 8: e01794.

Evans, A.M., R.G. Everett, S.L. Stephens, and J.A. Youlz. 2011. Comprehensive fuels treatment practices guide for mixed conifer forests: California, Central and Southern Rockies, and the Southwest.

Hallin, W.E. 1957. Silvical characteristics of Jeffrey pine. Technical Paper # 17. U.S. Department of Agriculture, Forest Service, California Forest and Range Experiment Station, Berkeley, CA.

Hollister, J.W. and T. Shah. 2017. elevatr: Access Elevation Data from Various APIs. R.

Larson, A.J., R.T. Belote, C.A. Cansler, S.A. Parks, and M.S. Dietz. 2013. Latent resilience in ponderosa pine forest: effects of resumed frequent fire. Ecological Applications 23:1243-1249.

Liaw, A., and M. Wiener. 2002. Classification and regression by randomForest. R News, 2(3): 18–22.

Lydersen, J.M., B.M. Collins, M. Coppoletta, M.R., Jaffe, H. Northrop, S.L. Stephens. 2019. Fuel dynamics and reburn severity following high severity fire in a Sierra Nevada, USA, mixed-conifer forest. Fire Ecology 15: 43

Metcalfe, P., K. Beven, and J. Freer. 2018. dynatopmodel: Implementation of the Dynamic TOPMODEL Hydrological Model. R.

Meyer, M.D., M.P. North, A.N. Gray, and H.S. Zald. 2007. Influence of soil thickness on stand characteristics in a Sierra Nevada mixed-conifer forest. Plant and Soil 294 (1–2): 113–123

Miller, J.D., and B. Quayle. 2015. Calibration and validation of immediate post-fire satellite-derived data to three severity metrics. Fire Ecology 11:12–30.

Miller, J., and A. Thode. 2007. Quantifying burn severity in a heterogeneous landscape with a relative version of the delta Normalized Burn Ratio (dNBR). Remote Sensing of Environment 109:66-80.

Minore, D. 1979. Comparative autecological characteristics of northwestern tree species—a literature review. Gen. Tech. Rep. PNW-GTR-087. Portland, OR: US Department of Agriculture, Forest Service, Pacific Northwest Research Station: 1-72.

Moody, T. J., J. Fites-Kaufman, and S.L. Stephens. 2006. Fire history and climate influences from forests in the northern Sierra Nevada, USA. Fire Ecology 2:115-141.

National Interagency Fire Center. 2019. Historical GeoMAC Perimeters 2012.

North, M. P., J.T. Stevens, D.F. Greene, M. Coppoletta, E.E. Knapp, et al. 2019. Tamm Review: Reforestation for resilience in dry western U.S. forests. Forest Ecology and Management 432: 209–224.

Parks, S., C. Miller, C. Nelson, and Z. Holden. 2014. Previous fires moderate burn severity of subsequent wildland fires in two large western US wilderness areas. Ecosystems 17:29-42.

R Core Team. 2019. A Language and Environment for Statistical Computing. R Found Stat Comput Vienna, Austria. Retrieved from https://www.r-project.org/

Ritchie, M.W., E.E. Knapp, and C.N. Skinner. 2013. Snag longevity and surface fuel accumulation following post-fire logging in a ponderosa pine dominated forest. Forest Ecology and Management 287:113-122.

Roussel, J., and D. Auty. 2019. lidR: Airborne LiDAR data manipulation and visualization for forestry applications. R.

Russell, W.H., J. McBride, and R. Rowntree. 1998. Revegetation after four stand-replacing fires in the Lake Tahoe Basin. Madrono 45:40-46.

Shive, K.L., H.K. Preisler, K.R. Welch, H.D. Safford, R.J. Butz, K.L. O'Hara, and S.L. Stephens. 2018. From the stand scale to the landscape scale: predicting the spatial patterns of forest regeneration after disturbance. Ecological Applications 28:1626–1639.

Stan Development Team. 2018. RStan: the R interface to Stan.

Thompson, J.R. and T.A. Spies. 2010. Factors associated with crown damage following recurring mixed-severity wildfires and post-fire management in southwestern Oregon. Landscape Ecology 25: 775-789.

USDA Forest Service 2016. Forest Service Activity Tracking System (FACTS). National Resource Manager, USDA Forest Service. Information online: http://www.fs.usda.gov/main/r5/landmanagement/gis

USDA Forest Service. 2009. Common stand exam field guide. Natural Resource Information System, Washington, DC. Available on-line at: http://www.fs.fed.us/nrm/fsveg/.

USDA Forest Service. 1989. Silvicultural Forest Handbook. U.S. Department of Agriculture, Forest Service, Pacific Southwest Region, San Francisco, California, USA.

Vehtari, A., A. Gelman, and J. Gabry. 2017. Practical Bayesian model evaluation using leave-one-out cross-validation and WAIC. Statistics and Computing 27:1413–1432

Westerling, A.L., B.P. Bryant, H.K. Preisler, T.P. Holmes, H.G. Hidalgo, T. Das, and S.R. Shrestha. 2011. Climate change and growth scenarios for California wildfire. Climatic Change 109: 445-463.

Ziegler, J.P., C.M. Hoffman, B.M. Collins, J.W. Long, C.M. Dagley and W. Mell. 2020. Simulated fire behavior and fine-scale forest structure following conifer removal in aspen-conifer forests in the Lake Tahoe Basin, USA. Fire 3(3)

Appendix A: Contact information for key project personnel

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Appendix B: List of Completed/Planned Products

Graduate thesis

Paudel, Asha. Vegetation dynamics at different spatio-temporal scales in frequently burned mixed-conifer forests, northern Sierra Nevada range, California. Department of Geosciences. Florida Atlantic University. PhD expected spring semester 2022.

Articles in peer-reviewed journals

Lydersen, J.M., Collins, B.M., Coppoletta, M., Jaffe, M.R., Northrop, H. Stephens, S.L. 2019. Fuel dynamics and reburn severity following high severity fire in a Sierra Nevada, USA, mixed-conifer forest. Fire Ecology 15: 43 (https://doi.org/10.1186/s42408-019-0060-x)

Steel, Z., Foster, D., Coppoletta, M., Lydersen, J.M., Wing, B., Stephens, S.L., Collins, B.M. Ecological resilience and vegetation transition in the face of multiple severe wildfires. In review.

Paudel, A., Coppoletta, M., Merriam, K., Markwith, S. Persistent suppression legacy, rapid change, and the absence of passive system restoration in Sierra Nevada mixed-severity reburns. In preparation.

Presentations

Markwith, S.H. The case of the missing fuels or beyond pre-European fire: complex cultural impacts on forest fuel loads. Presented at the 2020 Florida Society of Geographers Annual Meeting in Gainesville, FL.

Paudel, A. and Markwith, S.H. Mixed conifer density and regeneration variation in a mixed-severity fire landscape of the northern Sierra Nevada Range, California. Presented at the 2020 Florida Society of Geographers Annual Meeting in Gainesville, FL.

Markwith, S.H., Paudel, A., Coppoletta, M., Merriam, K., and Collins, B. Successive large mixed-severity wildfires and vegetation and fuel dynamics in the Sierra Nevada, CA. Presented in the Fire-Driven Vegetation Change session at the 2019 American Association of Geographers Annual Meeting in Washington, D.C.

Markwith, S.H., Paudel, A., Coppoletta, M., Merriam, K., and Collins, B. Analysis of vegetation change due to repeated large mixed-severity wildfires in the Sierra Nevada, CA. Presented at the 2019 Florida Society of Geographers Annual Meeting in Orlando, FL.

Paudel, A., Markwith, S.H., Coppoletta, M., Merriam, K., and Collins, B. Fuel and vegetation response to frequent fire at different spatio-temporal scales in the dry conifer forests, northern Sierra Nevada range, CA. Presented at the 2019 Florida Society of Geographers Annual Meeting in Orlando, FL.

Coppoletta, M. (annual). The role of fire in shaping forest ecosystems of the Sierra Nevada. Lecture and field day for UC Berkeley Forestry Camp students (June 2017, June 2018, July 2019)

Markwith, S.H., Paudel, A., Coppoletta, M., Merriam, K., and Collins, B. The effects of successive large mixed-severity wildfires on vegetation and fuels in the Sierra Nevada, CA. Presented at the 2018 Association of Pacific Coast Geographers Annual Meeting, Reno, Nevada.

Coppoletta, M., Merriam, K., and B. Collins. Influence of post-fire vegetation and fuels on fire severity patterns in reburns: implications for restoration. Presentation given to UC Davis Safford lab. January 25, 2017

Coppoletta, M., Merriam, K., and B. Collins. Influence of post-fire vegetation and fuels on fire severity patterns in reburns: implications for restoration. Presentation given at the 88th Annual Meeting of the Northwest Scientific Association. Southern Oregon University, Ashland, Oregon. March 29-31, 2017

Posters

Paudel, A., Coppoletta, M., Merriam, K, and Markwith, S.H. Vegetation dynamics in frequently burned mixed conifer forests, northern Sierra Nevada Range, CA. Presented at the 2020 Ecological Society of America Conference, August 3-6, 2020.

Markwith, S.H., Paudel, A., Coppoletta, M., and Merriam, K. Mixed conifer regeneration after successive large mixed-severity wildfires in the Sierra Nevada, CA. Presented at the 2019 Association for Fire Ecology 8th International Fire Ecology and Management Congress, November 18-22, 2019, Tucson, Arizona.

Paudel, A., Markwith, S.H., Coppoletta, M., Merriam, K., and Collins, B. Vegetation and fuel dynamics at different spatio-temporal scales in frequently burned mixed conifer forests, northern Sierra Nevada Range, California. Presented at the 2019 American Association of Geographers Annual Meeting in Washington, D.C.

Website

Our project website is hosted by the Department of Geosciences at Florida Atlantic University (http://www.geosciences.fau.edu/reburn/). This website features photos and brief descriptions of the project objectives, methodologies, and outcomes. This site is regularly updated and contains links to presentations, publications, and other deliverables as they become available.

Appendix C: Metadata

We utilized two types of data to address our research objectives: (1) data collected in conjunction with a prior project; and (2) new data collected as part of this JFSP project.

Data that fall under the first category include:

- Light Detection and Ranging (LiDAR) data and high-resolution aerial imagery collected in 2009 (after the initial fires) and LiDAR data collected in 2013 (after the 2012 reburn). These data are available upon request from the Pacific Southwest Region Remote Sensing Laboratory or the Plumas National Forest.
- Tree, fuels, vegetation, and ground cover measurements collected between 2009-2012 in common stand exam plots established within the Storrie and Rich fire footprints, prior to the Chips Fire reburn. Data from these plots are currently archived in the Forest Service Research Data Archive (http://dx.doi.org/10.2737/RDS-2015-0039).

Data that fall under the second category (new data) include:

- Quantitative plot-level tree, fuels, vegetation, and ground cover data collected during the remeasure of previously established field plots (described above), as well as newly established plots. All data collected as part of this JFSP project are being uploaded to the USDA Forest Service National Research Data Archive (https://www.fs.usda.gov/rds/archive/) by Michelle Coppoletta. Data will be available by January 2021.