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Post-fire vegetation and fuel development influences fire severity patterns in reburns

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Abstract. In areas where fire regimes and forest structure have been dramatically altered, there is increasing concern that contemporary fires have the potential to set forests on a positive feedback trajectory with successive reburns, one in which extensive stand-replacing fire could promote more stand-replacing fire. Our study utilized an extensive set of field plots established following four fires that occurred between 2000 and 2010 in the northern Sierra Nevada, California, USA that were subsequently reburned in 2012. The information obtained from these field plots allowed for a unique set of analyses investigating the effect of vegetation, fuels, topography, fire weather, and forest management on reburn severity. We also examined the influence of initial fire severity and time since initial fire on influential predictors of reburn severity. Our results suggest that high- to moderate-severity fire in the initial fires led to an increase in standing snags and shrub vegetation, which in combination with severe fire weather promoted high-severity fire effects in the subsequent reburn. Although fire behavior is largely driven by weather, our study demonstrates that post-fire vegetation composition and structure are also important drivers of reburn severity. In the face of changing climatic regimes and increases in extreme fire weather, these results may provide managers with options to create more fire- resilient ecosystems. In areas where frequent high-severity fire is undesirable, management activities such as thinning, prescribed fire, or managed wildland fire can be used to moderate fire behavior not only prior to initial fires, but also before subsequent reburns.

Key words: fire ecology; fire hazard; fire severity; interacting fires; post-fire restoration; reburn; Sierra Nevada, California, USA.

Introduction

Management of dry forests in the western United States is increasingly motivated by wildfire. Management activities associated with fire intend to mitigate impacts in advance of fire (fuels reduction and ecological restoration treatments), during fire (suppression), and following fire (reforestation and fuels reduction treatments). With increasing incidence of large fires in the western United States (Dennison et al. 2014) there is greater demand on public land managers to plan for and implement these mitigation activities. There is a strong basis demonstrating the need for fuels reduction and restoration in dry forests to promote fire resilience (Franklin and Johnson 2012, Stephens et al. 2013); the need for suppression actions to protect life, property, and other vulnerable resources is also widely recognized. The need for post-fire management actions, however, is less clear. It has been argued by some to let post-fire successional processes proceed without intervention (Donato et al. 2006, Swanson et al. 2011). However, given that the current extent of stand-replacing fire effects are thought to be outside the historical range of variability (Miller and Safford 2012, Mallek et al. 2013), some studies suggest post-fire management may be necessary to promote soil and water conservation, fire resilience, biodiversity, and re-establishment of pre-fire vegetation types, such as coniferous forest (Barton 2002, Goforth and Minnich 2008, Moreira et al. 2012, Collins and Roller 2013).

In addition to a focus on post-fire vegetation recovery, there is concern over fuel development following large fires. Fires consume fuel, which can limit the extent and effects of subsequent fires (Collins et al. 2009, Parks et al. 2014). However, where tree mortality is high, there is increased potential for hazardous surface fuel accumulations, as fire-killed trees lose limbs and ultimately fall (Keyser et al. 2009, Ritchie et al. 2013). This is exacerbated in most dry forest types of the western United States, where fire exclusion has led to very high pre-fire tree densities (Parsons and Debenedetti 1979, Naficy et al. 2010, Scholl and Taylor 2010, Collins et al. 2011, Stevens-Rumann et al. 2013). In addition to elevated fine

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and coarse woody debris in areas with high tree mortality, 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 an abundance of evidence from fire history studies demonstrating recurring fire on short return intervals (5–15 years) in dry forests throughout the western United States (e.g., Fulé et al. 1997, Heyerdahl et al. 2001, Brown et al. 2008, Taylor 2010). These fires predominantly burned at low to moderate severity, leaving the overstory component of these forests largely intact. Over time, these frequently occurring fires resulted in a stabilizing negative feedback (sensu Larson et al. 2013), in which fires consumed surface fuel and in some cases understory and mid-story vegetation (ladder fuels), reducing the intensity and effects of subsequent fires (Fig. 1). This feedback of low- to moderate-severity fire begetting future fires with similar effects exists in some contemporary forests where fire regimes have been restored (e.g., Collins et al. 2009, Lydersen and North 2012, Larson et al. 2013, Parks et al. 2014). However, this is not the case for much of the dry forest types in the western United States. Altered forest conditions resulting from decades of fire exclusion and overstory removal harvesting have changed contemporary fire patterns, leading to uncharacteristically large stand-replacing patches (Miller et al. 2012, Stephens et al. 2013). There is great concern that these altered patterns in contemporary fires, particularly large fires, will set forests on a positive feedback trajectory with subsequent fires; one in which extensive stand-replacing fire will promote more stand-replacing fire (Fig. 1; Thompson and Spies 2010). This type of feedback has the potential for long-term state change from forest to persistent shrubland or grassland (Collins and Skinner 2014). The fact that changing climatic conditions may be leading to greater large fire potential increases the likelihood for these

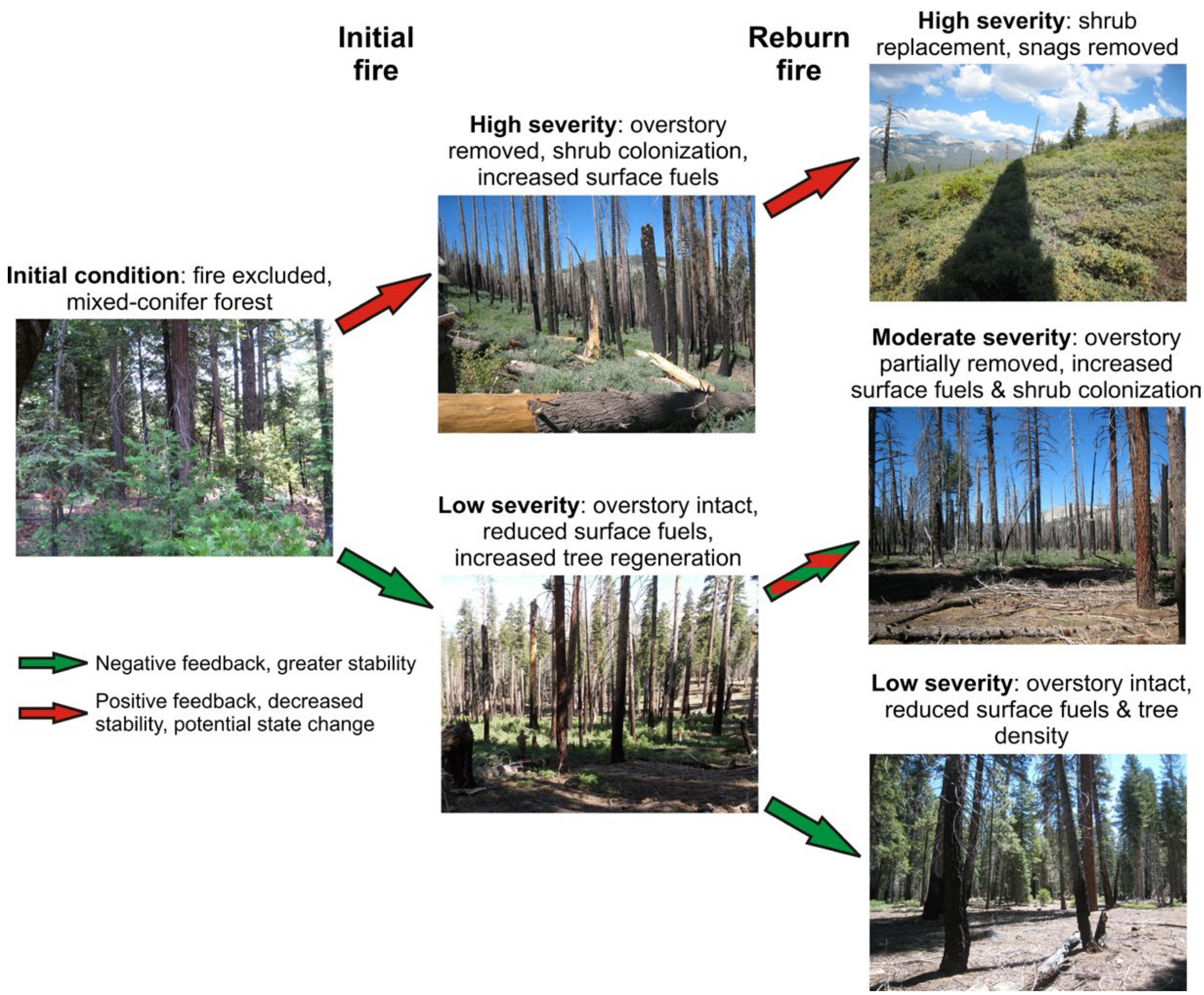

Fig. 1. Conceptual model of potential pathways for post-fire vegetation and fuel dynamics following initial fires and reburns. Time between initial fire and reburn is assumed to be relatively short (5–15 yr). Pathways are coded for the type of ecological feedback based on expected change to the dominant vegetation in response to different fire severity levels and effects these vegetation changes would have on subsequent fires.

long-term vegetation state changes (Westerling et al. 2011, Collins 2014, Dennison et al. 2014).

In this study, we use an extensive set of field plots established following four fires that occurred between 2000 and 2010 in the northern Sierra Nevada, California, USA. Portions of these four fires were reburned by a 30000-ha fire in 2012, which burned 118 previously established field plots. The information obtained from these field plots allowed for a unique set of analyses investigating the factors contributing to observed reburn severity. The specific questions we sought to answer were: (1) what was the effect of site-level biophysical characteristics, topography, post-fire management, and fire weather on reburn severity; (2) did initial fire severity and time since initial fire, either individually or in combination, influence reburn severity; and (3) if so, how did initial fire severity and/or time since initial fire, influence the variables identified as important drivers of reburn severity? By addressing these questions, we hoped to identify characteristics that managers could target to influence the potential severity and frequency of subsequent fires to create more fire-resilient forest landscapes.

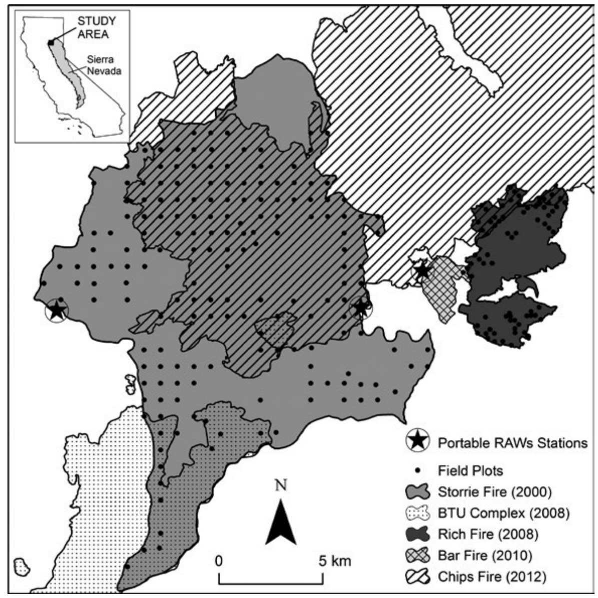

Fig. 2. Locations of field plots (n = 249) in the Storrie, Rich, BTU Complex, Bar, and Chips fires on the Plumas and Lassen National Forests, California, USA. Field plots were characterized by lower montane mixed-conifer forest prior to the four initial fires.

Methods

Study area

The study area encompasses ~25160 ha in the northern portion of the Sierra Nevada mountain range, California, USA. It is defined by five fires that burned over a 12-yr period, including four initial fires (2000 Storrie Fire, 2008 Butte Lightning [BTU] Complex, 2008 Rich Fire, and the 2010 Bar Fire) and one reburn (2012 Chips Fire), which occurred shortly after, and significantly overlapped, the initial fires (Fig. 2, Table 1). The Storrie Fire ignited on 17 August 2000 in the Feather River Canyon on the Plumas National Forest (NF) and quickly spread to encompass 22687 ha of private and National Forest System lands before it was controlled on 27 September 2000. In June 2008, lightning ignited a series of fires, collectively referred to as the BTU Complex, on the Plumas and Lassen National Forests. Prior to containment, two of these fires burned 2625 ha within the Storrie Fire perimeter. On 29 July 2008, the Rich Fire ignited just east of the Storrie Fire and burned 2473 ha of private and National Forest System lands before it was controlled on 9 August 2008. The Bar Fire began on 31 July 2010 and burned 421 ha, including a small portion (6.6 ha) of the Rich Fire. The Chips Fire ignited on 29 July 2012 within the Storrie Fire footprint and reburned ~45% (10166 ha) of the Storrie Fire area before moving northeast and reburning 379 ha within the Rich Fire perimeter. California fire perimeter data show that in the century preceding the four initial fires, less than 15% of the study area had experienced fire (perimeter data *available online*).

The five fires in the study area all occurred within the North Fork Feather River watershed, where the topography is generally mountainous and steep and slopes are predominantly southeast-facing. Elevations within the study area range from 525 to 2160 m. The area is characterized by a Mediterranean climate with warm, dry summers and cold, wet winters. Prior to the initial fires, vegetation was predominantly lower montane mixed-conifer forest, consisting of white fir (*Abies concolor*), Douglas fir (*Pseudotsuga menziesii*), ponderosa pine (*Pinus*

Table 1. Characteristics of the four initial fires sampled.

Fire name (year)	Total fire area (ha)	Number of plots	Time since initial fire (years)	Elevation (m)	Slope (degrees)
Storrie (2000)	22687	163	9–11	1550 (679–2053)	18 (2–44)
BTU Complex (2008)	2625†	13	2–3	1406 (923–1922)	22 (6–35)
Rich (2008)	2473	71	3–4	1551 (1025–1918)	19 (1–38)
Bar (2010)	421	2	1	1566 (1515–1616)	19 (19–20)

Note: Elevation and slope are mean values for all plots in a given fire with the range in parentheses.
†This area represents the overlap between the BTU Complex and the Storrie Fire.

ponderosa), Jeffrey pine (*Pinus jeffreyi*), sugar pine (*Pinus lambertiana*), incense-cedar (*Calocedrus decurrens*), and California black oak (*Quercus kelloggii*). Some stands of red fir (*Abies magnifica*) were found at higher elevations and on moister sites. Montane chaparral, characterized by wild lilac (*Ceanothus spp.*) and manzanita (*Arctostaphylos spp.*) species, was present throughout the study area, but dominated lower elevation, drier, and southeast-facing canyon slopes. Fire and vegetation management in this area is primarily focused on restoration of vegetation types that are adapted to frequent, low-severity fire regimes; therefore we limited our analysis to sites that were lower montane mixed-conifer forest prior to the four initial fires (n = 249).

Field data

We used data from two separate sampling efforts. The first effort was conducted within the 2000 Storrie Fire footprint between 2009 and 2011 and encompassed a portion of the 2008 BTU Complex. The second effort occurred within the 2008 Rich Fire footprint between 2011 and 2012 and encompassed a small portion of the 2010 Bar Fire. In both fires, field plots were established in a systematic grid across the fire area. Plot centers were established on the vertices of a 800-m grid in the Storrie Fire and a 200-m grid in the Rich Fire. Field data for both fires 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 2007). A larger plot is used to collect tree data (>12.7 cm diameter at breast height, dbh), including status (live or dead), dbh, and species. The common stand exam protocol allows for some variation in methodology. For example, in the Storrie Fire, a prism was used to establish a variable radius larger plot. In the Rich Fire this larger plot had a fixed 16-m radius. Sampling conducted 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. A total of 249 field plots were established in areas that were lower montane mixed-conifer forest prior to the four initial fires; 118 of these reburned in the 2012 Chips Fire (Fig. 2).

Vegetation composition was assessed by identifying all plants to the species level, and estimating percent cover for each species. These data were then aggregated by lifeform and status to obtain percent cover values for all trees, shrubs, and understory species (forb and graminoid) present in the plot. In the Storrie Fire, these data were collected within the 5-m radius plot, while in the Rich Fire these data were collected in the larger 16-m radius plot. Ground cover data was identified according to the following categories: rock, bare ground, woody debris, litter, basal vegetation, and other. In the Storrie Fire, ground cover was measured along a 6-m transect using a point intercept method, with points spaced at 6-cm intervals. In the Rich Fire, the percent cover of each ground surface category was visually estimated within the larger 16-m plot. Fuels data, including fine fuels, duff depths, and rotten and sound 1000-h fuels, were collected using a planar intersect technique (Brown 1974). In the Storrie Fire, one 10.7-m transect was used per plot; in the Rich Fire, four 9-m transects were established at each plot.

We recognize that differences in data collection methods between the two fires are an inherent limitation of these data. During our exploratory analysis, we utilized t tests to determine if significant differences between variables could be attributed to data collection method. We found no significant differences (two-tailed t test, $P > 0.05$) between data collection methods for any of the variables except mean snag density (snags/hectare) and percent cover of rock, litter, wood, and bare ground. We decided to include all of the variables in our final analysis because other factors, such as fire history or site conditions, may also have contributed to these observed differences. In addition, none of the five variables that differed between data collection methods were identified as strong predictors of reburn severity in our exploratory analysis.

Spatial data

Fire severity was estimated using the Relative differenced Normalized Burn Ratio (RdNBR), derived from Landsat Thematic Mapper data acquired during the first growing season after the fire was controlled (Miller and Thode 2007). RdNBR corrects for the correlation between pre-fire NBR and total biomass, and as a result, more robustly captures fire-caused change across the range of vegetation conditions (e.g., closed canopy forest, low-density forest, shrub-dominated; Miller and Thode 2007). Fire severity values for each plot were extracted from 30-m resolution raster data using bilinear interpolation. This sampling approach was necessary because plots rarely fell within the center of the pixels. Bilinear interpolation uses the value of the four nearest pixel centroids to calculate a weighted average, which is adjusted to account for the distance of the four cells from the plot center. To assess the performance of RdNBR as a metric of burn severity, we used linear regression to test the correlation between initial fire severity and the proportion of total tree basal area (live and dead) that were snags after the initial fires. The assumption with this assessment was that a majority of snags measured were living trees prior to the initial fire and were killed by the fire.

Weather variables were obtained from three portable Remote Automated Weather Stations set up to collect weather data during the Chips Fire (Fig. 2). Hourly values of temperature, relative humidity, and wind speed were averaged to get daytime (10:00–17:00) estimates of each variable; these values were then averaged over the three weather stations. Daily progression maps for the Chips Fire, produced by Forest Service staff, were used to assign daily average fire weather values to each plot based on the day that it burned. Topographic variables, which included elevation (m), slope (degrees), Beers' transformed aspect (Beers et al. 1966), and slope curvature (i.e., convex or concave), were derived for each plot from a 10-m digital elevation model. The Land Facet Corridor Designer ArcGIS extension developed by Jenness et al. (2013) was used to derive topographic position for each plot at two spatial scales using 300- and 1000-m circular neighborhoods (Thompson et al. 2007, Thompson and Spies 2009, 2010). Plots were assigned one of five topographic position classes: valley, lower slope, middle slope, upper slope, or ridge. Slope, aspect, slope curvature, and topographic position were then used to derive the Topographic Relative Moisture Index (TRMI) using the methods described in Parker (1982). Each plot was assigned a TRMI value that represented relative soil moisture availability and ranged from 0 (xeric) to 60 (mesic).

We used the Forest Service Activity Tracking System (USDA 2014) to identify management activities that occurred within plots between 2000 and 2013. Plots were considered treated if management activities occurred after data collection, but prior to reburning (n = 19). Management activities included tree planting (n = 15) and chipping, piling, and burning of fuels (n = 4). Due to the small diversity and number of activities, we categorized plots as either treated or untreated for analysis. In our analysis of initial fire severity effects, plots were excluded if they were treated prior to data collection (n = 20). Management activities that occurred in plots prior to data collection included thinning, salvage logging, and tree planting. Data collection in plots occurred 1–12 yr after the initial fires. Time since initial fire was derived for each plot by calculating the number of years between data collection and the most recent fire. The close proximity of the fires in the study area (Fig. 2) and their similar elevations, slopes, aspects, and vegetation types (Table 1) allowed us to analyze time since initial fire, independent of specific fire event. Forest Service vegetation maps (USDA Forest Service 2011) were used to determine the dominant vegetation type in each plot prior to the initial fires. Plots that were montane chaparral (n = 31), hardwood (n = 10), red fir (n = 7), or other non-forest vegetation types (n = 8) prior to the four initial fires were excluded from analysis.

Statistical Analyses

What was the effect of site-level biophysical characteristics, topography, post-fire management, and fire weather on reburn severity?.—For each of the 118 plots established prior to, and subsequently burned by the 2012 Chips Fire, we assembled a list of 24 variables to capture site-level biophysical characteristics, topography, post-fire management, and fire weather (Table 2). We used a two-stage approach to explore the influence of these variables on reburn severity. First, we used a random forest analysis (RF) to identify influential explanatory variables and rank their importance. We used the importance rankings to reduce the number of potential explanatory variables for the second stage, which was regression tree analysis (RT). By only including influential predictor variables identified from the RF we attempted to minimize potentially spurious results that RT can be prone to (Maloney et al. 2009). The intent of the RT was to identify specific thresholds for predictor variables that explained the observed variation in reburn severity. This two-step approach has been shown by others to be well suited for analyses of complex ecological data, particularly when there are numerous potential predictor variables, as well as hierarchical and nonlinear relationships among predictor variables and between predictor and response variables (e.g., Thompson and Spies 2010, Walter and Platt 2013, Lydersen et al. 2014). RF and RT analyses were performed using the party package (Hothorn et al. 2006, Strobl et al. 2007, 2008) developed for R (R Core Team 2013).

In our first step, we used RF to construct a suite of regression trees, using a randomly selected subset of predictor variables and a random subsample of data for each tree. Examining a large number of regression trees allows for identification and ranking of influential variables, and averages out the instability of individual regression trees that can exhibit large changes in structure due to random variation in the data (Strobl et al. 2009). We used the function cforest, which generates conditional inference trees that avoid the bias in variable selection that can be problematic with other regression tree methods (Strobl et al. 2007). The number of variables used in constructing each tree was five, following the standard practice of using the square root of the total number of predictor variables. Stable rankings of predictor variables were produced by running three replicate random forest analyses, each with 5000 regression trees. We used conditional permutation importance measures to rank predictor variables (Strobl et al. 2008). This method can be more accurate than traditional importance rankings, which tend to exaggerate the significance of variables that are highly correlated with influential predictors. Predictor variables with importance values greater than the absolute value of the lowest negative score were considered influential or of potential interest (Strobl et al. 2009).

In our second step, RT was performed using the conditional inference tree technique, which uses a partitioning algorithm based on the lowest statistically significant P value (α = 0.01) derived from Monte Carlo simulations. This method minimizes bias and prevents over-fitting of the data (Hothorn et al. 2006).

Did initial fire severity and time since initial fire, either individually or in combination, influence reburn severity?.—Using the same 118 plots that reburned during the Chips Fire, we explored the relationship between initial fire severity, time since initial fire, and reburn severity with generalized additive (GAM) and general linear (GLM) models. GAMs are extensions of GLMs that allow the data to drive the shape of the fitted curve through the inclusion of nonparametric smoothing functions; this

Table 2. Summary of response and predictor variables used in the random forest analysis.

Variable	Mean	Minimum	Maximum	Standard deviation
Response				
Chips Fire reburn severity (Relative differenced Normalized Burn Ratio)	284.2	−1145.0	933.0	293.7
Predictor				
Vegetation				
Tree cover (%)	27.5	0.0	89.0	25.4
Shrub cover (%)	24.6	0.0	100.0	31.0
Understory cover (%)	12.0	0.0	80.0	19.9
Live tree density (ha^{-1})	399.3	0.0	4693.0	664.0
Snag density (ha^{-1})	311.2	0.0	3577.7	489.0
Live tree basal area (m^2/ha)	34.5	0.0	126.4	30.9
Snag basal area (m^2/ha)	21.4	0.0	190.6	26.6
Dead and down fuels				
Total fine woody debris (Mg/ha)	17.9	0.0	120.5	20.6
1000 h rotten fuels (Mg/ha)	23.9	0.0	539.5	75.1
1000 h sound fuels (Mg/ha)	14.0	0.0	158.8	33.2
Duff (Mg/ha)	5.0	0.0	66.1	7.3
Ground cover				
Bare ground cover (%)	8.1	0.0	66.0	11.6
Basal vegetation cover (%)	6.9	0.0	87.0	16.1
Rock cover (%)	15.6	0.0	100.0	20.1
Litter cover (%)	52.5	0.0	96.0	26.2
Wood cover (%)	16.6	0.0	90.0	20.2
Topography				
Elevation (m)	1504.2	679.2	1970.2	309.5
Beers-transformed aspect	1.1	0.0	2.0	0.7
Slope (degrees)	18.2	3.0	38.0	8.8
TRMI (300 m)	29.9	4.0	53.0	9.8
TRMI (1000 m)	28.9	4.0	49.0	9.5
Daily fire weather				
Average temperature (°C)	28.1	22.8	31.0	2.3
Average relative humidity (%)	21.8	11.7	30.9	5.4
Average wind speed (km/h)	6.7	4.4	10.6	0.9

Notes: Categorical variable management activities was also taken into account; plots were considered treated if activities occurred after data collection, but prior to the reburn. 15.1% of plots (n = 19) were treated, 84.9% of plots (n = 99) were untreated.

flexibility often makes GAMs well suited for exploration of nonlinear relationships (Hastie and Tibshirani 1986, 1990, Wood 2006). Relationships were modeled using penalized thin-plate regression splines (Hastie and Tibshirani 1990) from the mgcv package (Wood 2006) in R (R Core Team 2013). As suggested by Hastie and Tibshirani (1990) and Yee and Mitchell (1991), we fitted parametric functions whenever inclusion of nonlinear effects did not significantly improve the fit of the model (assessed using Akaike information criterion; AIC) or when significant interactions were present. We used a normal (Gaussian) probability distribution and identity link function.

How did initial fire severity and/or time since initial fire, influence the variables identified as important drivers of reburn severity?.—To address our final question, we used GAM and GLM models to investigate the effect of initial fire severity and time since initial fire on the biophysical variables that were identified as important predictors of reburn severity in our RT model. In this analysis, we attempted to establish more robust relationships by using the full plot data set (249 plots), since we were not specifically investigating reburn effects. We excluded 21 plots from this analysis because Forest Service management activities occurred between the initial fire and data collection. Again, linear models were used whenever inclusion of nonparametric smoothing functions did not significantly improve the fit of the model (assessed using AIC) or when significant interactions were present. We used normal (Gaussian) probability distributions, log and identity link functions, and square root transformations whenever necessary to meet the assumptions of the models (see Table 3).

Results

Plots were distributed across a range of RdNBR values in both the initial fires and the reburn (Fig. 3). In both

Table 3. Summary of generalized additive (GAM) and general linear (GLM) models used to analyze the effect of initial fire severity and time since initial fire on reburn severity, shrub cover, and snag basal area.

Response variable	Best-fit model	R^2	Predictor variables	P
Reburn severity (RdNBR)	GLM; Gaussian distribution (identity link)	0.43	initial severity	0.029
			time since initial fire	0.378
			initial severity × time since initial fire	0.001
Shrub cover	GAM; Gaussian distribution (log link)	0.31	s(initial severity)^2	<0.001
			s(time since initial fire)^2	0.001
Snag basal area	GLM; Gaussian distribution (identity link)	0.14	initial severity	<0.001
			time since initial fire	0.002
			initial severity × time since initial fire	0.014

Notes: Models with the best fit are presented; s = nonparametric smoothing function used in GAMs; P-values for GAMs represent approximate significance of smooth terms. Snag basal area was transformed for analysis: square root (snag basal area +1).

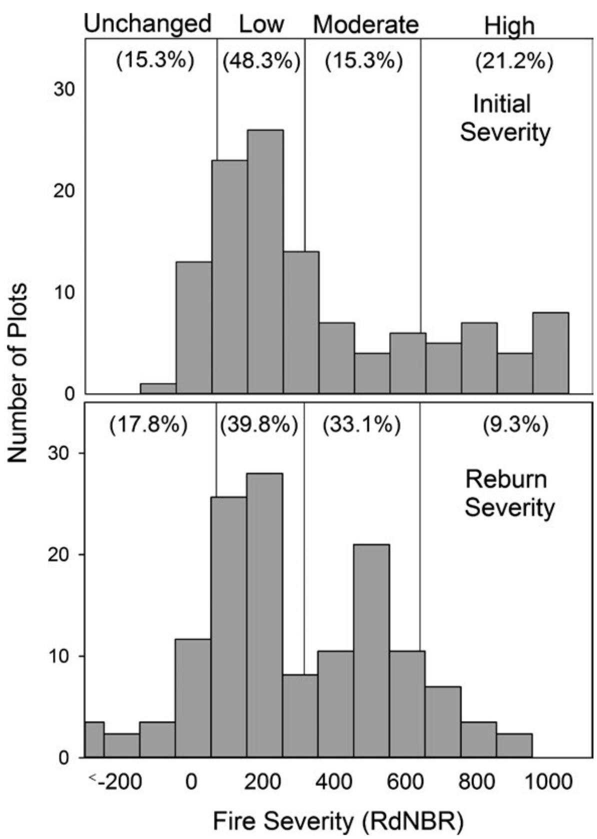

Fig. 3. Distribution of fire severity (RdNBR) among plots that burned in the initial fires and were reburned by the Chips Fire. The percentage of plots within each severity category is also included. Fire severity classes are based on Miller and Thode (2007).

cases, the highest percentage of plots fell within the range of values that generally correspond with low fire severity (RdNBR < 315). The number of plots that burned at high to moderate severity was slightly higher in the reburn (42.4%) compared to the initial burn (36.5%). RdNBR values from the initial fires were significantly correlated with the percent total basal area of snags (R^2 = 0.43, P < 0.001), which suggests that the use of RdNBR as a continuous metric of burn severity in our analysis is reasonable.

Effect of biophysical, topographic, post-fire management, and fire weather variables on reburn severity

The RF analysis identified 17 influential predictors of reburn severity (Fig. 4). Average daytime temperature was identified as the most important predictor. Average relative humidity and shrub cover were also highly associated with reburn severity, while snag basal area, fine woody fuels, and live tree basal area were moderately associated. Understory plant cover, average daily wind speed, percent cover of wood and basal vegetation, elevation, 1000-h rotten woody fuels, overstory tree cover, percent cover of bare ground, live tree density, percent cover of rock, and management activities were identified as marginal predictors of reburn severity. With the exception of elevation, topographic variables (aspect, slope, and TRMI), did not appear to substantially influence reburn severity.

The RT analysis explained 42% of the variance in reburn severity and produced a tree with four terminal nodes (Fig. 5). Average daytime temperature was the most important explanatory variable. Predicted reburn severity was lowest in plots that burned on days averaging less than or equal to 27.3°C. For plots that burned on days with higher average temperatures (>27.3°C), basal area of snags was the next most important explanatory variable, with plots containing more than 43 m^2/ha basal area of snags associated with the highest reburn severities. In plots with lower snag basal area (i.e., <43 m^2/ha), reburn severity was highest in plots that contained more than 60% cover of shrubs.

The influence of initial fire severity and time since initial fire on reburn severity

Initial fire severity and time since initial fire explained a significant amount of the variation in reburn severity (R^2 = 0.43, P = < 0.001). We found a significant interaction (P < 0.001) between initial fire severity and time

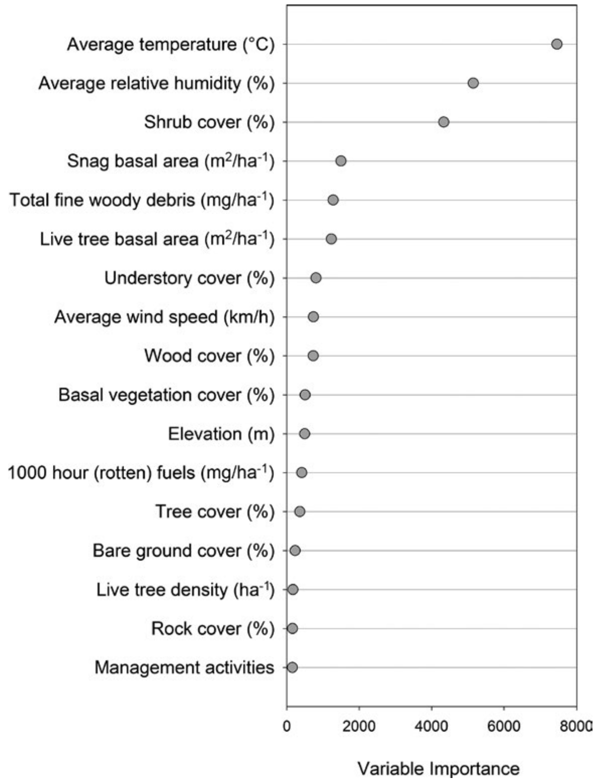

FIG. 4. Variable importance plot for the 17 influential predictor variables from the random forest model for reburn severity. Predictor variables are along the y-axis and the median conditional variable importance value is on the x-axis. Variables with importance values higher than the absolute value of the lowest negative score (x = 128), were considered potentially influential.

since initial fire (Table 3), indicating that the relationship between initial fire severity and reburn severity differed in relation to the amount of time that had elapsed since the initial fire. In older fires (e.g., those that burned more than 9 yr prior to data collection), plots that initially burned at higher severities tended to reburn at higher severities, whereas areas that initially burned at lower severities generally reburned at lower severities (Table 3, Fig. 6). This relationship was less apparent in younger fires (e.g., areas that reburned within 4 yr or less).

The influence of initial fire severity and time since initial fire on important drivers of reburn severity

Initial fire severity and time since initial fire had a significant effect on post-fire shrub cover (R^2 = 0.31, P < 0.001) and snag basal area (R^2 = 0.14, P < 0.001) (Table 3), the two most important biophysical predictors of reburn severity identified in our RT analysis (Fig. 5). The relationship between initial fire severity and post-fire shrub cover was nonlinear (Fig. 7). In areas that burned initially at moderate to high severity (RdNBR values > 316), post-fire shrub cover increased positively in response to increasing severity. In contrast, areas that burned at lower severities (RdNBR values 69–316) or were unchanged by the initial fire (RdNBR values <69), exhibited a negative relationship between initial severity and post-fire shrub cover. Time since initial fire had a positive linear effect on post-fire shrub cover (Table 3, Fig. 7). Snag basal area was positively and linearly related to initial fire severity; however this relationship varied in relation to time since initial fire. The effect of initial fire severity on snag basal area was much more apparent in the younger fires (Fig. 8).

DISCUSSION

Fire is an important ecological process in the Sierra Nevada, and has a strong influence on vegetation structure and species composition patterns across landscapes. These patterns, in turn, influence the behavior and severity of subsequent fires (McKenzie et al. 2011). Historically, frequent low-severity fire in Sierra Nevada mixed-conifer forests maintained low tree densities and limited the accumulation of live and dead fuel, which reduced extensive crown fire (Show and Kotok 1924). Some studies have shown that despite decades without fire, the reintroduction of fire can be sufficient to restore historical forest conditions in mixed-conifer forests (e.g., Taylor 2010, Collins et al. 2011). However, these findings are mostly limited to areas that have relatively intact vegetation structure and composition (i.e., limited or no direct manipulation from timber harvesting). In most of the Sierra Nevada, where both fire regimes and forest structure have been dramatically altered, there is a greater likelihood that wildfire may be reinforcing, rather than restoring, these altered conditions (Thompson and Spies 2010, Crotteau et al. 2013).

Our results demonstrating a positive relationship between initial fire severity and the severity of a subsequent reburn are in agreement with findings from several previous studies (e.g., Holden et al. 2010, Thompson and Spies 2010, van Wagtendonk et al. 2012, Parks et al. 2014). However, our investigation differs from these previous studies because we incorporated field-based, quantitative measures of vegetation and fuels into our analysis, which allowed for identification of the potential mechanisms driving observed patterns of reburn severity. In our study, fire weather clearly had a strong influence on reburn severity (Fig. 4), but the air temperature threshold associated with greater reburn severity (27.3°C; Fig. 5) is not particularly extreme during the summer in this area. When this relatively common air temperature was exceeded, two of the most influential variables driving reburn severity patterns were snag basal area and shrub cover, both of which were influenced by initial fire severity and time since initial fire (Table 3, Fig. 5). Other studies have documented greater snag basal area (Cocke et al. 2005) and higher shrub cover (Crotteau et al. 2013, Cocking et al. 2014) in areas that have burned at moderate to high severity. While the outcomes from the initial fires are not surprising, the specific connection between initial fire severity, snag basal area, and shrub cover, and

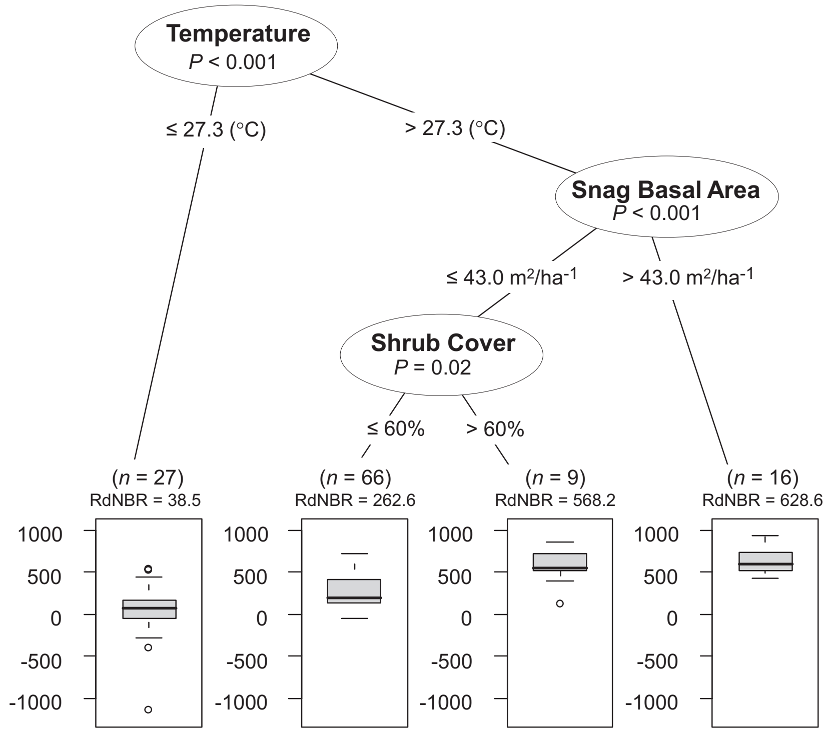

Fig. 5. Regression tree for reburn severity (RdNBR) using the 17 significant predictor variables identified in the random forest analysis. Values at the ends of terminal nodes represent the predicted reburn severity (mean RdNBR) and the number of plots in each group. P values at each node are from a Monte Carlo randomization test. The total R^2 for the tree is 0.43.

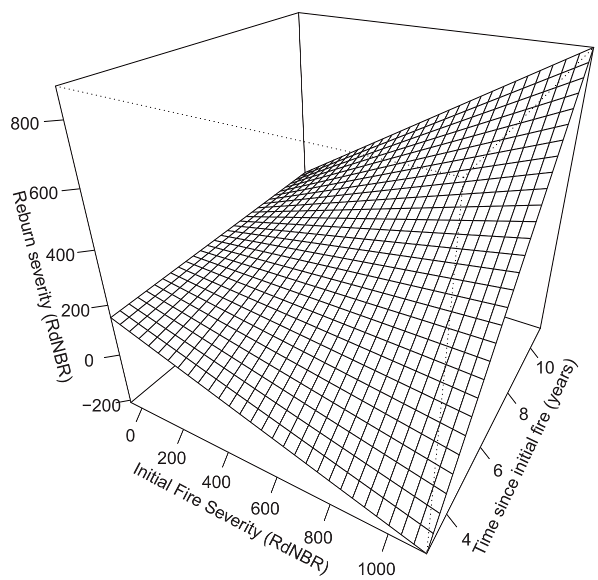

Fig. 6. Three-dimensional surface plot showing the interactive effect of initial fire severity and time since initial fire on reburn severity. Initial fire severity had a significant positive effect on reburn severity in older fires (i.e., those that burned more than 9 yr after the initial fire) but had no significant effect in younger fires (i.e., those that burned less than 4 yr after the initial fire).

the severity of a reburn, provides a greater mechanistic understanding of fire severity patterns. Our results suggest that initial high- to moderate-severity fire led to an increase in standing snags and shrub vegetation, which in combination with severe fire weather, promoted high-severity fire in the subsequent reburn.

Standing snags affect fire behavior and spread by acting as both a source for embers and a receptive surface for ignition from embers (van Wagtendonk 2006). As standing snags fall, they contribute to elevated surface fuel loads (McIver and Ottmar 2007, Ritchie et al. 2013), hinder fire suppression efforts (Brown et al. 2003), increase fire residence time (Brown et al. 2003, Monsanto and Agee 2008), and indirectly result in torching of live trees as a result of preheating (Ritchie et al. 2013). Given that a majority of our plots initially burned 12 years prior to reburning, it is likely that many of the snags were decayed, which increases their flammability once they become ground fuels (Stevens-Rumann et al. 2012). Interestingly, the threshold at which snag basal area became strongly associated with high-severity reburn in our analysis (43 m^2/ha) exceeded the live tree threshold (e.g., 34.4 m^2/ha for pine–mixed conifer) often used by forest managers as an indicator of greater susceptibility to insect infestation (Sartwell and Stevens 1975). This finding suggests that forest conditions prior to the initial fires may have ultimately influenced reburn severity by promoting the conversion of dense pre-fire forest stands to dense post-fire standing snags.

The extent of shrub vegetation dramatically increased as a result of high-severity fire during the initial fires. Comparisons between 1997 and 2009 vegetation maps

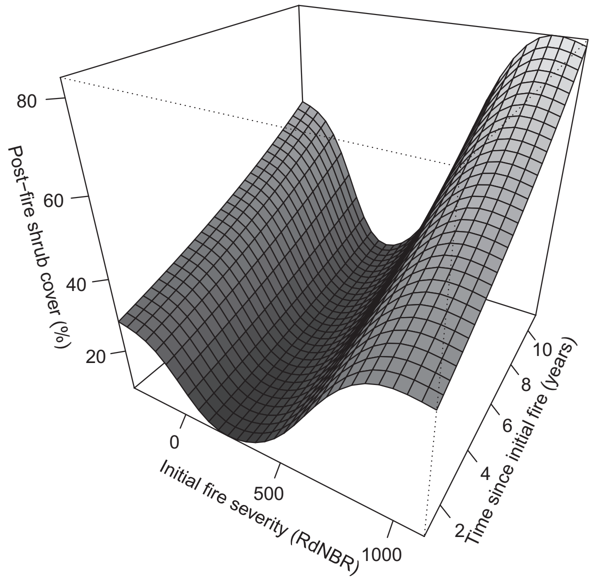

Fig. 7. Predicted values of post-fire shrub cover as a function of initial fire severity and time since initial fire using a generalized additive model.

(USDA Forest Service 2011) suggest that ~34% of lower montane mixed-conifer dominated plots were converted to montane chaparral after the 2000 Storrie Fire and 2008 BTU Complex (USDA Forest Service 2011). Historically, shrubs were a common component of Sierra Nevada mixed-conifer forests, but had a very patchy distribution (Knapp et al. 2012). This hetereogeneous occurrence of shrubs within a matrix of overstory trees serves as important habitat for many species of wildlife (White et al. 2013) and contributes to greater fine-scale heterogeneity across the landscape (Nagel and Taylor 2005). Large shrub patches, however, have the potential to homogenize landscapes (Stephens et al. 2013).

Unlike other studies of fire severity (e.g., Dillon et al. 2011, Birch et al. 2015), we found that topography was not a strong predictor of reburn severity. This was somewhat surprising given the known influence topography has both directly (e.g., slope steepness) and indirectly (e.g., aspect) on fire behavior. There are several potential explanations for the lack of statistical influence in our study. With regard to the lack of direct topographic influence, it is possible that the scale at which we summarized topographic variability, which was the local neighborhood around plots, was not the appropriate scale to capture its influence on fire behavior. A larger observational unit may have revealed a stronger topographic influence on fire severity patterns; however that would have required detailed vegetation and fuels data that are generally not available for areas larger than individual field plots (0.05–0.1 ha). The lack of indirect topographic influence may have been masked by our inclusion of detailed field-based measures of fuels and vegetation into our analysis and the short interval between successive fires. Topographic variables like elevation, slope, and aspect influence vegetation composition and structure, as well as the accumulation and spatial arrangement of fuels (Dillon et al. 2011). For example, southern aspects generally receive more solar radiation and have lower moisture availability; these conditions can result in drier fuels, smaller trees, and more shrub vegetation, all of which create conditions that can promote higher burn severity (Alexander et al. 2006).

An increase in reburn fire severity with time since initial fire has been demonstrated previously (Parks et al. 2014). A common explanation for this effect is that time since initial fire is a proxy for fuel accumulation (Collins et al. 2009), and longer periods of time result in higher live and dead fuel loads, which can lead to greater heat output from combustion and higher fire severity (Lydersen et al. 2014, Parks et al. 2014). The fact that older fires (i.e., areas reburned more than 9 years after the initial fire) were more likely to reburn at higher severity than younger fires (i.e., those that reburned less than 4 years after the initial fire) (Fig. 6) is consistent with other studies that have shown that fuels can recover to their pre-burn levels in 9–15 years (Thompson and Spies 2009, van Wagtendonk and Moore 2010). In our study, the relationship between snag basal area and initial fire severity was less apparent in the older fires, which may be an indicator of snag fall and ground fuel accumulation. Donato et al. (2013) also found that the window of low reburn potential may close relatively quickly (5–10 years) as regenerating vegetation and litter accumulates on the surface.

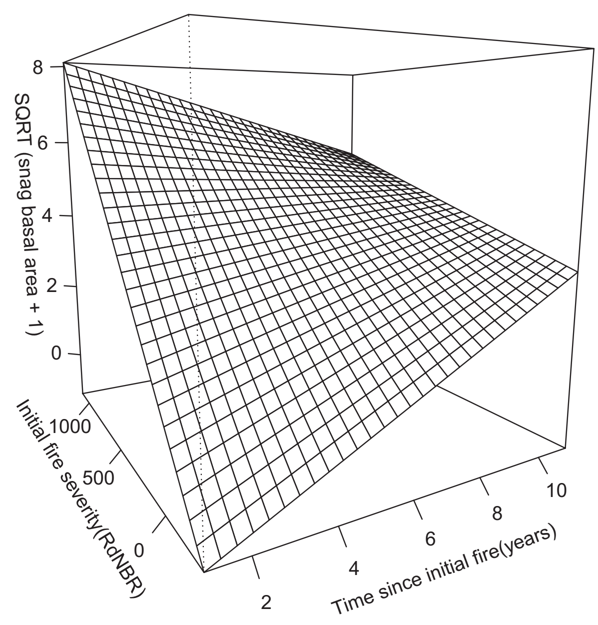

Fig. 8. Three-dimensional surface plot showing the interactive effect of initial fire severity and time since initial fire on snag basal area. Snag basal area was transformed for analysis using the square root (snag basal area +1). Initial fire severity had a significant positive effect on snag basal area in younger fires (i.e., those that burned less than 4 yr after the initial fire) but had no significant effect in older fires (i.e., those that burned more than 9 yr after the initial fire).

A strong shrub response following high-severity fire can be a desirable outcome, particularly in areas where montane chaparral has been lost due to fire suppression (Nagel and Taylor 2005, Cocking et al. 2014). In fact, early seral plant communities that develop following high-severity fire provide habitat to a number of wildlife species (e.g., Swanson et al. 2011, Seavy et al. 2012). The question is, what level of fire-initiated early seral vegetation is desirable? More specifically, what proportion of the landscape should be in the early seral stage and how large should patches be? While there may be no clear answer to these questions, several studies have demonstrated that large shrub patches (e.g., >100 ha) can considerably slow or inhibit the development of pine-dominated forests (Barton 2002, Goforth and Minnich 2008, Roccaforte et al. 2012, Collins and Roller 2013, Crotteau et al. 2013). Although we do not have post-reburn data on tree regeneration, it is reasonable to assume that conifer seedlings that may have established in high-severity patches after the initial fires were killed in areas that reburned at high severity. Many shrub species have traits, such as resprouting, that allow them to persist in areas that experience repeated high-severity fires. Conifers in this forest type lack such traits and can only establish when seed from adjacent mature, live trees is dispersed by the wind or small mammals. These different post-fire establishment mechanisms, coupled with the propensity for areas with high shrub cover to reburn at high severity (Fig. 5) suggest the likelihood for a positive fire severity feedback loop that promotes persistent montane chaparral (Fig. 1, Thompson and Spies 2010).

Our findings indicate that areas burned initially at low and moderate severity tended to reburn at similar severities (Fig. 6). Low- to moderate-severity effects in contemporary fires have been shown to approximate historical forest structure and patterns of spatial heterogeneity (Taylor 2010, Lydersen and North 2012). Additionally, it has been demonstrated that moderate-severity fires may produce surface fuel loads that fall within a more optimal range, i.e., one that balances wildlife and understory productivity with fire hazard (Brown et al. 2003, Stevens-Rumann et al. 2012). Collectively, these studies and our findings suggest that there is potential for contemporary fires, even when reburning in short succession, to increase resilience. In our study, this potential was higher when reburning took place under less extreme weather conditions (Fig. 5), which has also been demonstrated in other studies (Lydersen et al. 2014).

Management implications

Fire behavior is largely driven by weather variables, including temperature and relative humidity. In the face of changing climatic regimes and increases in extreme fire weather, managers may feel they have few options to restore low- and moderate-severity fire regimes. However, our data show that components of vegetation composition and structure are important drivers of fire severity in both initial and subsequent fires. Factors such as vegetation and fuels not only contribute to patterns of repeated high severity but can also be manipulated through management action. Altering conditions so that disturbance processes can act to increase, rather than reduce, forest heterogeneity may provide ecosystems with the ecological flexibility (Holling 1973) to withstand and persist through future changes in climate and climate-related processes (Safford et al. 2012a).

In areas where frequent, high-severity fire is undesirable, management activities can focus on moderating fire behavior not only prior to the initial fires, but also before subsequent reburns. Properly implemented fuel treatments have been well documented to moderate the behavior of wildfires (Agee and Skinner 2005, Raymond and Peterson 2005, Safford et al. 2009, 2012b, Vaillant et al. 2009). The results of our study suggest that fuels treatments that reduce stand density may also reduce the density of post-fire snags and the risk of high-severity reburns. Unfortunately, the pace and scale of treatments needed to effectively reduce fuels across large landscapes often surpasses the implementation capacity of many land management agencies (North et al. 2012). Land allocations (e.g., wilderness areas, inventoried roadless areas) and feasibility constraints (e.g., distance from roads, steep slopes) also limit the area available for mechanical fuel treatments. Given these restrictions, fuel reduction treatments will need to be strategic, both in their placement on the landscape (e.g., to create fuel breaks) and purpose (e.g., to protect refugia conifer populations or other highly valued habitats). Where and when conditions allow, fuel reduction activities may also include the managed use of wildland fire under moderate weather conditions (Steel et al. 2015).

Post-fire management actions should be tailored to fire severity and long-term management objectives, with a focus on enhancing resilience to future reburns. In high-severity areas where snag densities are high, selective removal may be an effective method to reduce the accumulation of coarse woody ground fuels over time (Stevens-Rumann et al. 2012). In some cases, post-fire thinning can increase surface fuels by relocating tops, branches, and broken boles to the ground during harvest (Thompson et al. 2007). To avoid these effects, thinning treatments need to be followed up with additional fuel reduction activities, such as broadcast-burning or piling and burning (Thompson et al. 2007). There are a number of studies that have outlined the potential negative impacts of post-fire logging (Beschta et al. 2004, Donato et al. 2006) and these risks need to be balanced with the risk of future high-severity fire. For example, thinning may not be necessary in some areas where post-fire fuel loads fall within an acceptable range or are considered ecologically beneficial (Brown et al. 2003). Post-fire treatments that are focused on fuel reduction, rather than cost-recovery, may consider retention of larger legacy snags, if considered beneficial for wildlife, and focus removal efforts on dead biomass trees.

Prescribed fire and managed wildland fire can be an effective method for reducing fuel loads in both burned and unburned landscapes (Brown et al. 2003). Burning under controlled conditions provides managers with some level of control over the extent and amount of fuel consumption. Stevens-Rumann et al. (2012) suggest burning post-fire landscapes when 1000-h fuel moistures are high, so that there is some consumption of coarse woody debris and a reduction in detrimental soil heating. The positive benefits associated with prescribed fire treatments and managed wildland fire need to be balanced with potential negative impacts, including difficulty in controlling fire intensity, which can result in possible fire escape, smoke, unexpected soil damage, excessive consumption of coarse woody debris, unacceptable levels of tree mortality, or failure to achieve fuels objectives (Passovoy and Fulé 2006). In addition, if forest regeneration is a management goal, existing seedlings may need to be protected or avoided during prescribed fire treatments. Management actions designed to protect refugia populations of mature conifers may also be warranted to ensure seed sources for future regeneration.

Passovoy and Fulé (2006) suggest that passive management may be appropriate in places where snags and coarse woody debris do not exceed management thresholds and where salvage actions are not warranted for other reasons, such as public safety, insect infestation, or economic constraints. From a management perspective, the optimum range of down wood in fire-prone forests is one that balances the ecological functions of dead wood with its contribution to future fire potential (Brown et al. 2003). Some post-fire areas that are structurally intact may still respond to fire in a functional way. For example, our analysis suggests that low- to moderate-severity burns can be attained, even under severe fire weather conditions, as long as fuels and shrub vegetation are at lower densities and cover.

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Data Availability

Data associated with this paper have been deposited in the USDA Forest Service Research Data Archive: http://dx.doi.org/10.2737/RDS-2015-0039