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RESEARCH ARTICLE

Stand-replacing patches within a 'mixed severity' fire regime: quantitative characterization using recent fires in a long-established natural fire area

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Abstract The complexity inherent in variable, or mixed-severity fire regimes makes quantitative characterization of important fire regime attributes (e.g., proportion of landscape burned at different severities, size and distribution of stand-replacing patches) difficult. As a result, there is ambiguity associated with the term 'mixed-severity'. We address this ambiguity through spatial analysis of two recent wildland fires in upper elevation mixed-conifer forests that occurred in an area with over 30 years of relatively freely-burning natural fires. We take advantage of robust estimates of fire severity and detailed spatial datasets to investigate patterns and controls on stand-replacing patches within these fires. Stand-replacing patches made up 15% of the total burned area between the two fires, which consisted of many small patches (<4 ha) and few large patches (>60 ha). Smaller stand-replacing patches were generally associated with shrub-dominated (*Arctostaphylos* spp. and *Ceanothus* spp.) and pine-dominated vegetation types, while larger stand-replacing patches tended to occur in more shade-tolerant, fir-dominated types. Additionally, in shrub-dominated types stand-replacing patches were often constrained to the underlying patch of vegetation, which for the shrub type were smaller across the two fire areas than vegetation patches for all other dominant vegetation types. For white and red fir forest types we found little evidence of vegetation patch constraint on the extent of stand-replacing patches. The patch dynamics we identified can be used to inform management strategies for landscapes in similar forest types.

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Introduction

Fire is an inherently complex landscape process, both within individual fires and among multiple fires over time (Romme 1982; Turner and Romme 1994; Agee 1998). This complexity is driven by heterogeneity in vegetation/fuels, topography, and local weather for individual fires, and by variability in timing, effects, and extents across multiple fires (Pyne et al. 1996). Many fire/landscape ecology studies seek to quantitatively resolve this complexity with the intent of explaining observed patterns for both recent fires (e.g., Turner et al. 1994; Collins et al. 2007; Roman-

Cuesta et al. 2009) and historical fires (e.g., Taylor and Skinner 1998; Brown et al. 1999; Taylor and Skinner 2003; Hessburg et al. 2005; Hessburg et al. 2007; Brown et al. 2008). Based on these studies our understanding of more natural fire/landscape dynamics has advanced and become more nuanced. Among the advancements, the importance of stand-replacing patches within a matrix of low to moderate fire-induced effects, or mixed severity, is being increasingly recognized (Hessburg et al. 2007; Beaty and Taylor 2008; Brown et al. 2008). While studies of historical fire characterize landscape patterns resulting from more natural fire–forest interactions, they are limited by availability of data. Specifically, such studies lack the information needed for detailed spatial analysis (e.g., actual fire extents, and range of fire effects).

The spatial information available for many recent fires has allowed for explicit spatial analysis of fires (Thompson et al. 2007; Roman-Cuesta et al. 2009). The problem with studying recent fires it that due to extensive fire exclusion, fire-induced effects, or fire severity, in recent fires can be exacerbated relative to more natural fire–forest interactions (Safford et al. 2008; Miller et al. 2009a). In a few areas throughout the western U.S. lightning-ignited fires have been allowed to burn for several decades (Rollins et al. 2002; van Wagtendonk 2007). Due to repeated burning these areas have as close to natural fire regime as any place in the western U.S (Collins and Stephens 2007). As a result, studying spatial characteristics of fires in these natural fire areas provides insight on the patterns and controls of more natural fire–forest interactions (Collins et al. 2007). Such studies can complement previous characterizations of historical fire–forest interactions and contribute towards better understanding of mixed-severity fire regimes. Furthermore, studying more recent 'natural' fire regimes may be particularly relevant given the potential differences between historical and projected future climates.

In this study we take advantage of an existing dataset, consisting of remotely-sensed fire severity estimates, to investigate patterns and controls on stand-replacing patches within recent natural fires. We do this for two large fires that occurred in the Illilouette Creek basin, Yosemite National Park, USA: 2001 Hoover fire (2144 ha) and 2004 Meadow fire (2141 ha). We limit this analysis to these two fires because they are the only fires that occurred in this long-standing natural fire area that post-date the development of a detailed vegetation map for the area, which occurred in 1997, and have existing fire severity assessments. By having detailed vegetation mapping prior to each of the fires we can robustly investigate the role of vegetation in influencing stand-replacing fire. Previous work in the Illilouette Creek basin focused on both the patterns of fire severity, based on analysis of individual pixels (Collins et al. 2007), and on interactions among fires (Collins et al. 2009). This work concentrates on stand-replacing patches created by fires and attempts to explain the controls operating on patch occurrence. Ultimately, we intend for this work to provide managers with information on managing, and possibly mimicking the complex process of fire across landscapes.

Methods

Study area

Illilouette Creek basin is located within Yosemite National Park, which is situated in the central Sierra Nevada, California, USA (Fig. 1). The basin is over 15,000 ha with elevations ranging from 1400 to nearly 3000 m for the surrounding ridges. The climate is Mediterranean with cool, moist winters, and warm, generally dry summers. Average January minimum temperatures range from −2 to 5°C, while average July maximum temperatures range from 19 to 31°C (1992–2009, Crane Flat Remote Automated Weather Station—RAWS). Precipitation varies with elevation and is predominantly snow, with an annual average of 62 cm (1993–2008, Crane Flat RAWS).

The upper elevation mixed-conifer forests of the Illilouette Creek basin are dominated by Jeffrey pine (*Pinus jeffreyi*), white fir (*Abies concolor*), red fir (*Abies magnifica*), and lodgepole pine (*Pinus contorta* var. *murrayana*), and are interspersed with meadows and shrublands. Shrublands are largely dominated by *Arctostaphylos* spp. and *Ceanothus* spp. Based on tree-ring reconstructions, the historical fire regime within Jeffrey pine-dominated stands predominantly consisted of frequent low- to moderate-severity fires. Collins and Stephens (2007) reported a mean fire interval of 6.3 years for the area, and a fire rotation of 24.7 years from 1700 to

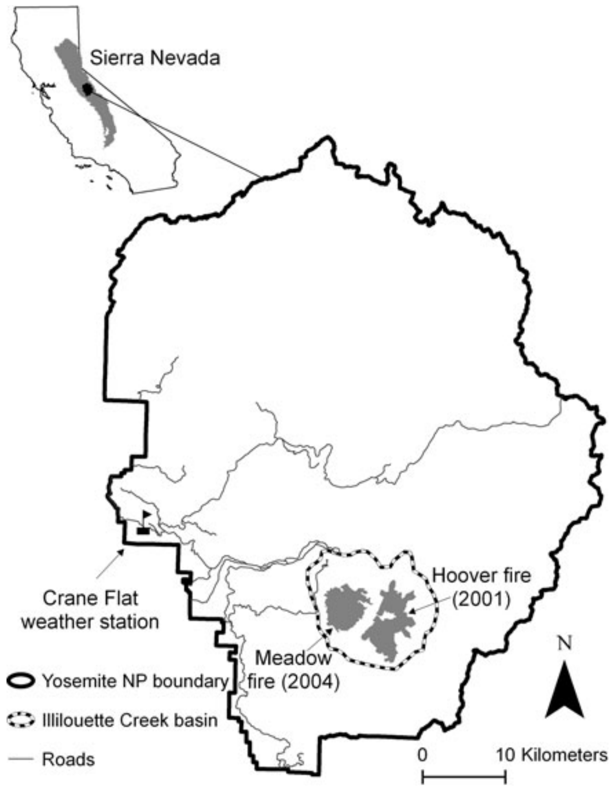

Fig. 1 Locations of the two studied fires within the long-established natural fire area, Illilouette Creek Basin, Yosemite National Park, California, USA

1900. The same study also found that the mean fire return interval and the fire rotation in the recent WFU period (1973–2005) did not differ noticeably from historical estimates (6.8 and 32.9 year, respectively).

Spatial data/analysis

We obtained perimeters and satellite-derived estimates of fire severity for both the Hoover (2001) and Meadow (2004) fires from the USDA Forest Service, Pacific Southwest Region Fire and Aviation Management personnel (J. Miller, personal communication 2007). These estimates are based on the relative differenced Normalized Burn Ratio (RdNBR), which is computed from Landsat TM imagery (see Miller and Thode 2007; and Miller et al. 2009b for more specific explanation of methodologies). This index has been used extensively to characterize relatively recent fires (Safford et al. 2008; Collins et al. 2009) and fire regimes (Holden et al. 2007; van Wagtendonk and Lutz 2007; Miller et al. 2009a). Miller et al. (2009b) report user's and producer's accuracies for RdNBR-based high severity classification to range between 70.7 and 85.3%, indicating RdNBR robustly captures high-severity fire effects.

Using classified RdNBR images (threshold values established by Miller and Thode 2007) we employed the patch delineation algorithm PatchMorph (Girvetz and Greco 2007) to identify contiguous patches of high-severity, or stand-replacing fire. The PatchMorph tool can be used in ArcGIS, and allows for users to specify maximum gap widths contained within individual patches, as well as minimum patch sizes and thicknesses. For this analysis we chose a maximum gap thickness of 60 m, or two 30 m pixels, and a minimum patch size of 0.5 ha. Lower gap thicknesses resulted in highly complex shapes containing several 'holes' within individual patches, while larger gap thicknesses failed to capture inherent heterogeneity. Our minimum patch size of 0.5 ha matches the minimum mapping unit for the vegetation map we used in the analysis.

For each stand-replacing patch identified using the PatchMorph tool we assembled a suite of variables that characterize the pre-fire vegetation, topography, previous fire history, and weather during burning. We used these as independent variables in statistical analysis (explained in following sub-section). The vegetation map we used was created in 1997, which pre-dates both fires. Based on the vegetation classes identified in this map we assigned a dominant vegetation type to each stand-replacing patch. The vegetation types we assigned were simplified based on dominant vegetation in order to have a reasonable number of classes for comparison among types. This simplification resulted in six dominant vegetation types: (1) shrub (predominantly *Arctostaphylos* spp. and *Ceanothus* spp.), (2) Jeffrey pine/shrub, (3) Jeffrey pine/white fir, (4) lodgepole pine, (5) white fir/red fir/lodgepole pine, and (6) white fir/red fir. In several instances stand-replacing patches consisted of multiple dominant vegetation types (e.g., see Fig. 3). In such cases we assigned the dominant vegetation type that occupied the largest proportional area. For topography we calculated the following variables: mean elevation, mean slope steepness, dominant aspect (4 categories), and slope position. These were all derived from a 10 m digital elevation model. We assigned slope position by first calculating the topographic position index for each 10 m pixel within our study area, using a 500 m focal mean, then classifying each pixel based on classes outlined by Jenness (2007).

Previous fire history was assessed using digitized fire atlases, which included all fires that occurred in the Illilouette Creek basin since the mid-1930's, although no significant fires existed in the database prior to 1974. We determined first whether or not each patch burned previously, and if so, how many years between that previous fire and each of the studied fires. Several patches had no record of being burned in the digital fire atlases, and as a result we had no estimate of time since last fire. As such, we had to use a categorical variable to characterize previous fire history, with the categories being: <17 yr, 17–30 yr, and >30 yr. The 17 year cutoff corresponds with previous work done by Collins et al. (2007) identifying the influence of time since last fire on observed patterns of fire severity, while the 30 year cutoff corresponds to the beginning of the WFU period. Time since previous fire for studied patches ranged from 10 to 30 yr.

We used two fire danger indices, burning index (BI) and energy release component (ERC), to capture weather conditions during burning (Bradshaw et al. 1984). Both indices are strongly influenced by weather, however they also incorporate influences of fuel structure and abundance on fire intensity (Bradshaw et al. 1984). Because the fuel component of BI and ERC calculations are based on relatively coarse-scale estimates of fuel structure and abundance, or fuel model, we use only one fuel model for the entire Illilouette Creek basin (NFDRS fuel model H—short-needle, normal dead). As such, we use BI and ERC as proxies for fire weather. BI and ERC values were derived from hourly temperature, wind speed, and relative humidity observations taken at Crane Flat RAWS. This station is approximately 30 km from the Illilouette basin and 600 m lower in elevation (see Fig. 1). We used the progression maps, which identify fire perimeters at various intervals throughout the duration of the fire, to apply BI and ERC values to each of the stand-replacing patches (Collins et al. 2007). We averaged hourly BI and ERC values to match the temporal extent of the perimeters in the progression maps. This extent was often daily during highly active burning periods, and up to several days or weeks during less active burning periods.

To investigate potential vegetation influences on stand-replacing patches we developed a metric to quantify the similarity in dominant vegetation types both within and immediately outside each patch. To do this we constructed a 100 m ring buffer around

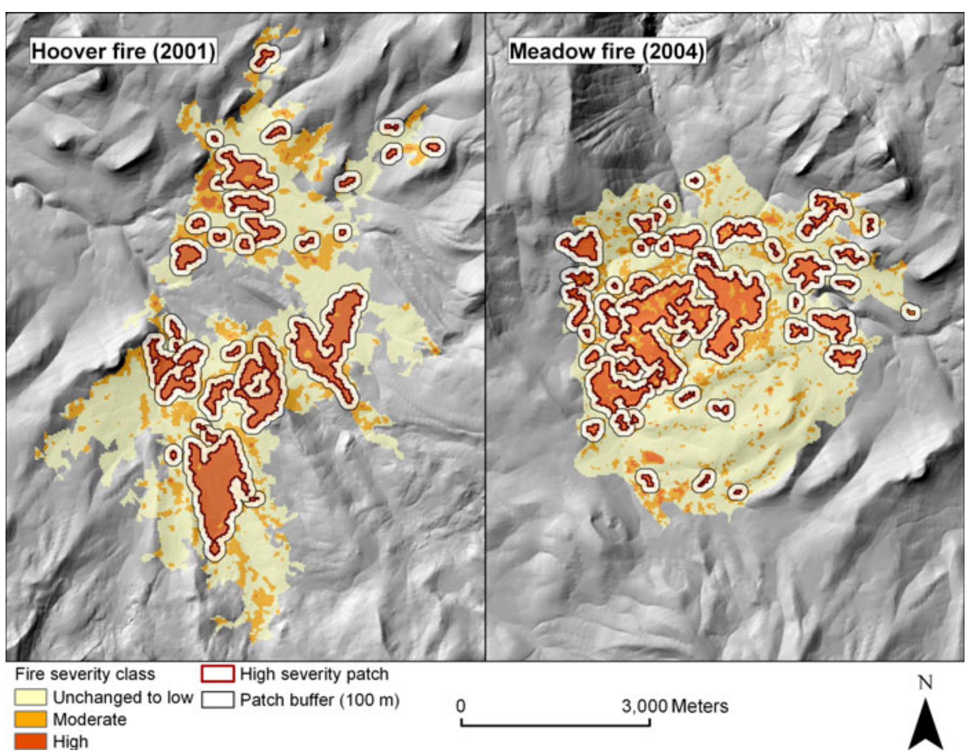

Fig. 2 Stand-replacing patches within the two studied fires based on the Landsat TM-derived relative differenced Normalized Burn Ratio (RdNBR, Miller and Thode 2007). Patches were delineated using the PatchMorph tool for ArcGIS (Girvetz and Greco 2007); see methods section for delineation criteria. Stand-replacing patches accounted for 15% of the combined area for both fires

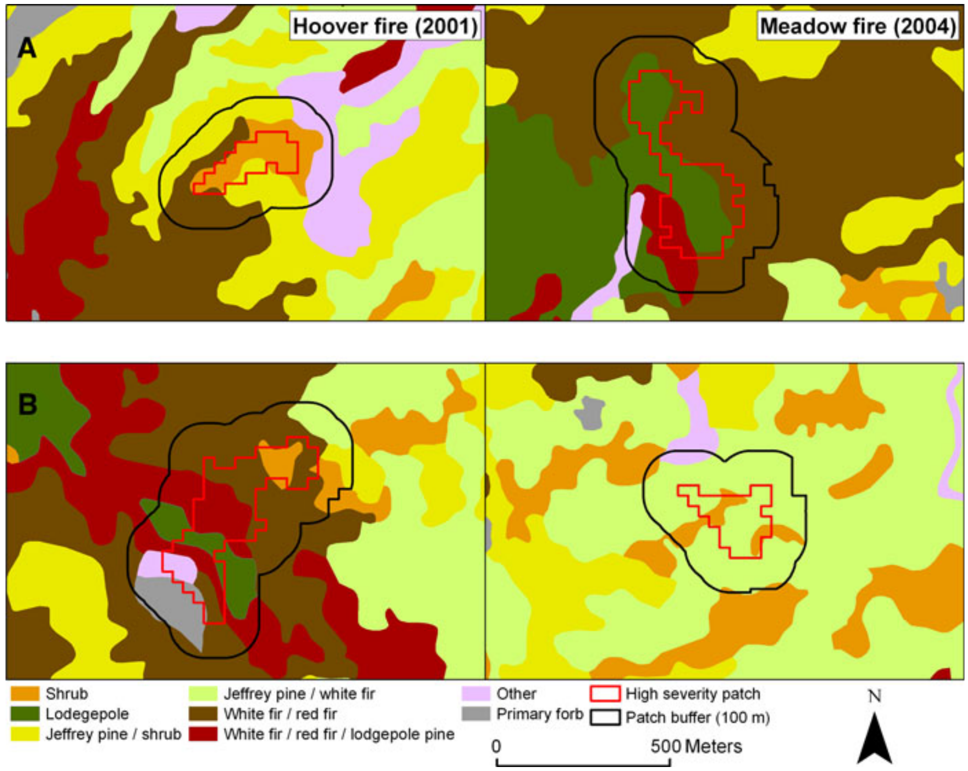

Fig. 3 Examples of both vegetation-constrained stand-replacing patches (**a**), and non-vegetation-constrained stand-replacing patches (**b**) within the two studied fires

each identified stand-replacing patch (Fig. 2). We then used distance and area measuring tools in ArcGIS to estimate the proportion of vegetation within the 100 m ring buffer that was similar to the dominant vegetation within the stand-replacing patch. The metric was expressed as a percent, rounded to the nearest 5%, and ranged from 0 to 100%. A low percent of similar vegetation within the buffer area would indicate the stand-replacing patch was fairly constrained to the vegetation patch; hence the vegetation had a strong influence on the observed high-severity fire effects (Fig. 3a). Conversely, a high percent of similar dominant vegetation within the buffer area would indicate that high-severity fire was not constrained by particular vegetation patch or patches (Fig. 3b), suggesting other factors were driving the observed high severity (e.g., local weather, topography, etc.).

In addition to delineating stand-replacing patches, we used the PatchMorph tool to delineate vegetation patches that intersected the two fires. We used our simplified dominant vegetation types for this patch delineation, with the exception that Jeffrey pine/shrub and Jeffrey pine/white fir patches were combined into one Jeffrey pine group due to a low number of observations in the Jeffrey pine/shrub group. We delineated vegetation patches in order to obtain estimates of underlying vegetation patch sizes and potential variability among dominant vegetation types. We also computed the proportion of area occupied by each of the dominant vegetation types for the Hoover and Meadow fire areas combined and for just the stand-replacing patches combined.

Statistical analyses

We used the two metrics: stand-replacing patch size and percent similar vegetation within 100 m buffer (BufferVeg), to quantify landscape—stand-replacing patch dynamics and function as dependent variables for statistical analysis. We explored possible relationships between these two dependent variables and each of the independent variables mentioned previously (dominant vegetation type, mean elevation, mean slope, dominant aspect, topographic position, reburn class, BI, and ERC) using regression tree analysis. Regression tree analysis offers clear advantages over traditional linear models because it can handle nonlinear or discontinuous relationships between variables, and high-order interactions (De'ath and Fabricius 2000). In addition, the hierarchical structure and identification of potential threshold values for independent variables is well suited for explaining ecological phenomena (De'ath and Fabricius 2000; Collins et al. 2009). The regression tree is constructed by repeatedly splitting the data into increasingly homogenous groups based on the dependent variable. Each split minimizes the sum of squares within the resulting groups. The number of terminal nodes was determined using the one-standard error rule on the cross-validated relative error (Breiman et al. 1984; De'ath 2002). We ran multiple iterations using this method to confirm the chosen number of terminal nodes.

We examined diagnostic plots of both stand-replacing patch size distributions and vegetation patch size distributions for each dominant vegetation type ($n = 5$) to determine whether or not the data were normally distributed. These plots revealed skewed or heavy-tailed distributions; transformations did little to improve distributions. As such, we employed non-parametric approaches to compare stand-replacing patch sizes and vegetation patch sizes among dominant vegetation types. Kruskal–Wallis tests (Ott and Longnecker 2001) indicated a difference in both stand-replacing patch sizes (Chi-squared = 12.39, df = 4, $P = 0.015$), and vegetation patch sizes (Chi-squared = 59.43, df = 4, $P < 0.0001$) among vegetation types. We proceeded with individual Wilcoxon tests for all possible pairwise comparisons among vegetation types ($n = 10$), using a Bonferroni-corrected α-level (0.005) to identify significant differences (Schwilk et al. 1997).

Results

Using the PatchMorph tool, with our criteria of 60 m maximum gap thickness and 0.5 ha minimum patch size, 77 individual stand-replacing patches were identified among both fires. We eliminated five patches from analysis, four of which corresponded with patches of bare rock and/or sparse vegetation, and one of which existed outside the progression map, and therefore could not be assigned weather variables. Of the remaining 72 patches the sizes ranged from 0.54 to 93 ha, with a mean patch size of 9.1 ha and a median patch size of 2.2 ha. Patch sizes followed an approximately Pareto distribution, with small patches ($\leq$4 ha) accounting for more than 60% of the total number of patches (Fig. 4). However, the few large patches (> 60 ha) accounted for almost half of the total stand-replacing patch area (Fig. 4). Stand-replacing patches accounted for approximately 15% of the burned area for each of the two fires (Table 1), as well as 15% of the total area for both fires (Fig. 2).

Regression tree analysis explained a fairly low proportion of the variance in stand-replacing patch sizes ($R^2 = 0.26$). Of the variance explained by the model, dominant vegetation type was most influential (Fig. 5). Stand-replacing patches dominated by shrubs, as well as those dominated by lodgepole pine and Jeffrey pine tended to be smaller, while patches dominated by white fir and red fir tended to be larger (Fig. 5). Within patches dominated by shrubs, lodgepole pine, and Jeffrey pine, slope position appears to have influenced patch size, with the smallest patches occurring on upper slopes, lower slopes, and valleys (Fig. 5). Stand-replacing patches that occurred on mid-slope and flat (<10% slope steepness)

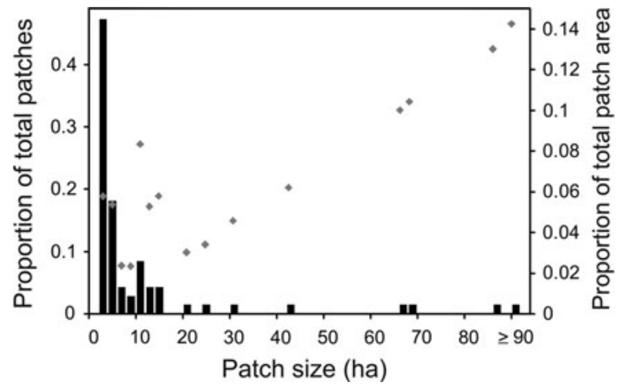

Fig. 4 Frequency distribution of stand-replacing patch sizes (black bars) and proportion of total stand-replacing patch area by size class (gray diamonds) within the Hoover (2001) and Meadow (2004) fires using a minimum patch size of 0.5 ha. There were a total number of 72 patches used in our analysis

Table 1 Area burned for two studied fires within the Illilouette Creek basin (Yosemite NP) summarized by fire severity class using the relative differenced Normalized Burn Ratio (Miller and Thode 2007)

	Hoover fire (2001)		Meadow fire (2004)	
	Hectares	%	Hectares	%
Unchanged to low	1152	54	1215	57
Moderate	667	31	595	28
High	325	15	331	15
Totals	2144		2141	

topographic positions tended to be larger. Within patches occurring on mid slopes and flatter areas the regression tree output identified further distinction in patch sizes based on the reburn class. Patches that burned between 17 and 30 years prior to either the 2001 Hoover fire or the 2004 Meadow fire tended to be larger than those that burned more recently or those that had no history of fire recorded in the digital fire perimeters (>30 yr) (Fig. 5).

Within the white and red fir dominated stand-replacing patches the regression tree analysis identified BI as having influenced patch size (Fig. 5). This split explained a high relative proportion of the total sum of squares, as indicated by the length of the 'branch', or line extending to the node, in the regression tree (Fig. 5). Fir-dominated patches that burned under periods of lower BI tended to be very small, averaging 1.3 ha, while patches that burned under periods of elevated BI tended to be much larger, averaging 25.4 ha.

A separate examination of stand-replacing patch size by dominant vegetation type revealed that patches dominated by white fir with lodgepole pine interspersed tended to be the largest (Fig. 6). Patches dominated by white fir and red fir had a tremendous range in patch size with an average of 15.1 ha, but a median of 1.8 ha. There were three large patches in white/red fir type (65, 68, and 93 ha) that caused this large discrepancy in mean and median patch sizes. In the shrub-dominated vegetation type, as well as in lodgepole- and Jeffrey pine-dominated types, differences between mean and median stand-replacing patch sizes were much smaller. Additionally, mean stand-replacing patch sizes for these vegetation types were noticeably smaller than those for fir-dominated groups (Fig. 6). Stand-replacing patch size distributions for the shrub-dominated, lodgepole-dominated, and Jeffrey pine-dominated types were significantly smaller that that for the white fir-lodgepole pine group. No other pairwise comparisons were significant based on the Bonferroni-adjusted α-level.

Analysis of all vegetation patch sizes that intersected the two fires revealed important differences among dominant vegetation types (Fig. 6). Shrub patches were smallest across the two fire areas, averaging approx. 4 ha. The distribution of shrub patch sizes was significantly different from that for all other vegetation types based on pairwise comparisons. No other pairwise comparisons were significant given Bonferroni-adjusted α-level; in large part due to the high variance in patch sizes in the two fir-dominated groups. The Jeffrey pine combined group (Jeffrey pine/shrub and Jeffrey pine/white fir) made up the largest proportion (0.34) of burned area for the two fires combined (Fig. 7). However, if both fir-dominated types were combined, fir would account for a greater proportion (0.45) of the burned area (Fig. 7). Shrub-dominated areas accounted for the

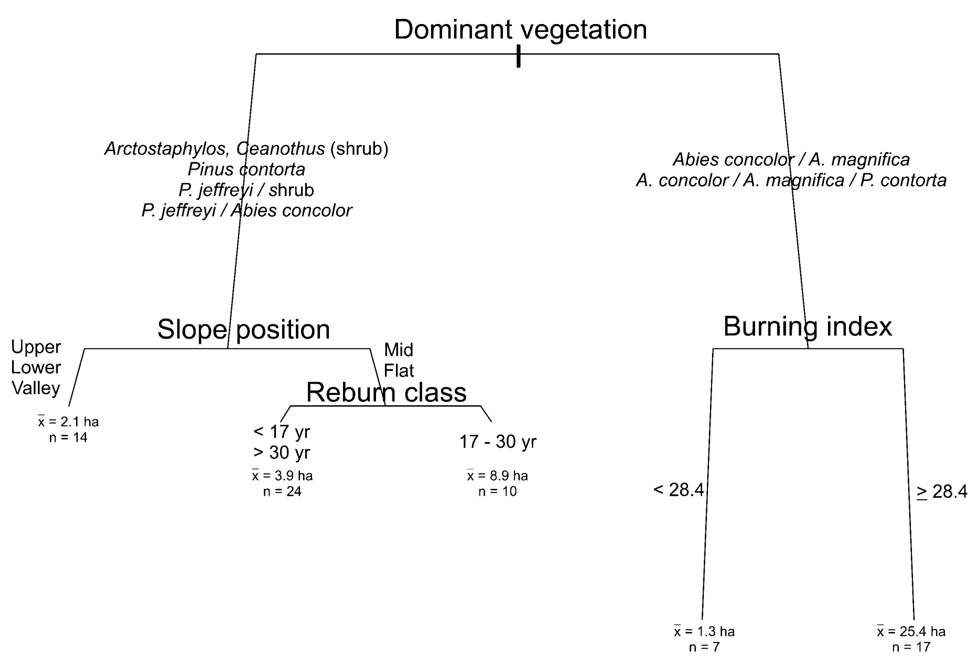

Fig. 5 Regression tree output explaining the influence of listed variables on stand-replacing patch sizes within the Hoover (2001) and Meadow (2004) fires. Average patch size ($\bar{x}$) and number of observations (n) are reported for the resulting group at each terminal node. The length of the line from each split indicates the relative proportion of total sum of squares explained by that split. The total R^2 for the tree is 0.26

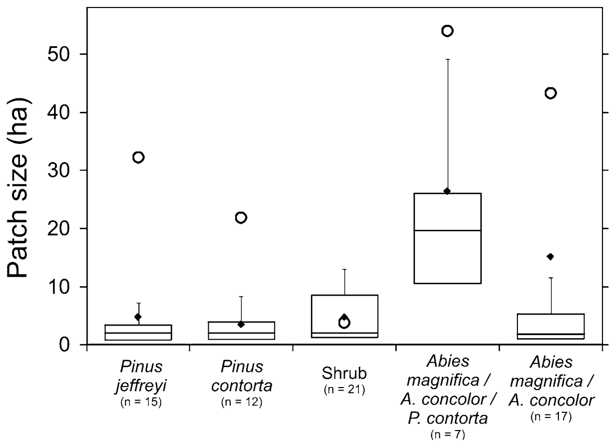

Fig. 6 Stand-replacing patch size box and whisker plots for each dominant vegetation type. Filled diamonds represent mean stand-replacing patch sizes for each vegetation type. Open circles represent mean vegetation patch sizes for all patches intersecting the two fire areas (Hoover 2001, Meadow 2004). Patches dominated by *Pinus jeffreyi*/shrub and *P. jeffreyi*/*Abies concolor* types were combined into one *P. jeffreyi* group due to a low number of observations in the *P. jeffreyi*/shrub group

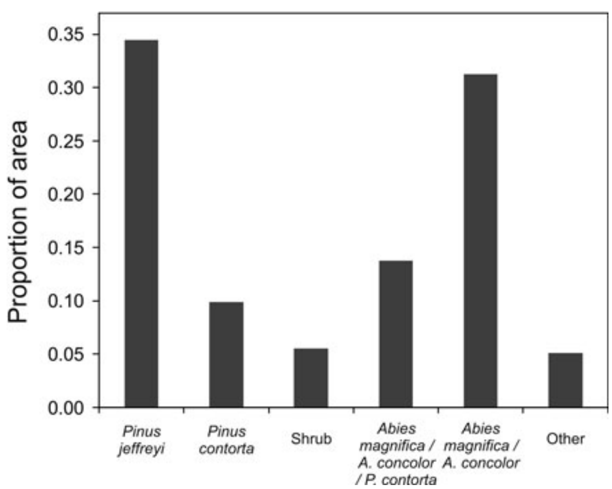

Fig. 7 Proportion of area within each dominant vegetation type summarized for the Hoover (2001) and Meadow (2004) fires combined (4285 ha)

smallest proportion of burned area (0.05) among our five dominant vegetation types. The proportion of burned area in the lodgepole type was 0.10 (Fig. 7).

Regression tree analysis explained a high proportion of the variance observed for our derived variable, percent similar vegetation within 100 m buffer (BufferVeg) ($R^2 = 0.83$). Dominant vegetation type accounted for much of the explained variance, with shrub-dominated stand-replacing patches having the lowest BufferVeg values (Fig. 8). Mean BufferVeg for shrub-dominated stand-replacing patches was 8.1%. Recall that lower BufferVeg values are indicative of more vegetation constraint on a given stand-replacing patch. Lodgepole pine and Jeffrey-pine/shrub types had intermediate BufferVeg values, averaging 60.3% between the two types (Fig. 8). White fir/red fir, white fir/red fir/lodgepole pine, and Jeffrey pine/white fir vegetation types had the highest BufferVeg values. Within these patches where fir was a dominant or co-dominant, the regression tree analysis identified slope steepness as having influence on BufferVeg values. Higher BufferVeg values were associated with slopes $\geq$14.4%, indicating that the least amount of vegetation constraint on stand-replacing patches existed in fir-dominant or co-dominant types on steeper slopes (Fig. 8).

Discussion

The imprecise definition of a 'mixed-severity' fire-regime leads to uncertainty in its use. Brown et al. (2008) make the point that at some spatial scale most fires are variable in severity, ranging from individual trees to large patches (>100 ha) of trees killed by crown fire. In the case of the fires we studied in the Illilouette Creek basin, we indeed detected a large range of high-severity, or stand-replacing, patch sizes. While it is clear that patches were generally small ($\leq$4 ha), large contiguous stand-replacing patches did occur (Figs. 2 and 4). The fact that these patches in total accounted for 15% of the burned area demonstrates that although stand-replacing fire is not the dominant process operating in these upper-elevation mixed-conifer forests, it is an important component of the fire regime.

Both the regression tree analysis of stand-replacing patch sizes (Fig. 5) and the pairwise comparisons of stand-replacing patch sizes between vegetation types (Fig. 6) identified a separation among conifer vegetation types. This separation indicates that larger stand-replacing patches tended to occur in fir-dominated vegetation types while smaller patches tended to occur in pine-dominated types (Figs. 5 and 6). Given that proportions of the total burned area are relatively similar between pine groups combined and the fir groups combined (Fig. 7), and that we found no differences in vegetation patch sizes between

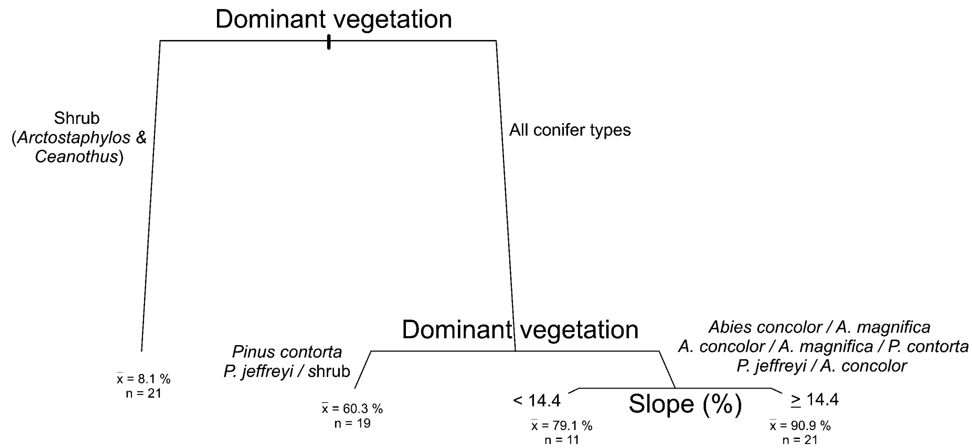

Fig. 8 Regression tree output explaining the influence of listed variables on vegetation—stand-replacing patch similarity within the Hoover (2001) and Meadow (2004) fires. As an indicator of potential vegetation constraint on stand-replacing patches we developed a variable estimating the percent of similar vegetation in a 100 m buffer surrounding each stand-replacing patch, relative the vegetation within the patch. This was the dependent variable for the regression tree analysis. Average percent similar vegetation ($\bar{x}$) and number of observations (n) are reported for the resulting group at each terminal node. The length of the line from each split indicates the relative proportion of total sum of squares explained by that split. The total R^2 for the tree is 0.83

these groups, this separation in stand-replacing patch sizes may not simply reflect the configuration/abundance of underlying vegetation. Vegetation structure within these dominant forest types may explain the difference in observed stand-replacing patch sizes. Fir-dominated forest stands tend to have multiple canopy strata (Beaty and Taylor 2008), resulting in greater vertical continuity in fuels from the ground surface to the upper canopy strata. This vertical continuity is largely due to the shade-tolerance of both white and red fir, and leads to stands being more susceptible to crown fire (Parsons and Debenedetti 1979; Scott and Reinhardt 2001). We speculate that stands in which fir makes up a large proportion of the mid-canopy and understory trees also have higher horizontal continuity (van Wagtendonk and Fites-Kaufman 2006). When combined with the increased susceptibility to crown fire, this horizontal continuity could lead to larger stand-replacing patches. In pine-dominated forest stands within the Illilouette basin, where fire frequency over the last 30+ years has not differed noticeably from historical fire frequency (Collins and Stephens 2007), we hypothesize there may be greater vertical and horizontal heterogeneity in forest structure (Stephens and Fry 2005; Stephens and Gill 2005). This heterogeneity may limit the extent of both passive and active crown fire (Brown et al. 2008; Stephens et al. 2008; North et al. 2009), resulting in smaller stand-replacing patches.

The finding indicating that smaller stand-replacing patches were also associated with shrub-dominated vegetation likely has a different explanation. The much smaller vegetation patches in the shrub-dominated type (Fig. 6) and the low proportion burned area occupied by the shrub type (Fig. 7) suggests that the smaller stand-replacing patches are likely a product of the underlying vegetation configuration and abundance. A confounding factor resulting from our analysis is that due to their small size shrub-dominated stand-replacing patches may tend to be underrepresented, especially in instances where there are many large patches. These larger patches would be associated with dominant vegetation types that represent greater proportions of the landscape (e.g. fir-dominated types). However, after visual inspection the number of instances in which shrub vegetation patches were contained within larger stand-replacing patches assigned to other dominant vegetation types were well below the number of distinct shrub-dominated stand-replacing patches (9 shrub patches contained within non-shrub stand-replacing patches, 21 distinct shrub stand-replacing patches).

Not surprisingly, higher burning index values, which indicate more severe fire weather, corresponded with the largest stand-replacing patches. More severe fire weather leads to greater fire intensity, which increases the potential for both lethal surface fire and crown fire (Albini 1976; Scott and

Reinhardt 2001). The difference in mean patch sizes between the two groups resulting from the split in burning index (means = 1.3 and 25.4 ha, Fig. 5), along with the high relative proportion of variance explained, suggest this relationship between fire weather and patch size is robust.

Previous work has demonstrated greater proportions of high-severity fire on upper slopes and lesser proportions on lower slopes (Beaty and Taylor 2008). Based on this work we would expect to see larger patches on upper slopes and smaller patches on lower slopes. Our analysis partially supports this assertion, in that smaller stand-replacing patches tended to occur on lower slopes and in valleys (Fig. 5). However, our results also indicate that smaller patch sizes tended to occur on upper slopes as well, which appears to contradict Beaty and Taylor (2008). When we overlayed areas classified as upper slopes onto digitized air photos it was apparent that our classification of upper slopes largely consisted of a relatively narrow band, approximately 100–300 m below the ridgetops. In many instances, these areas contained rock outcrops and/or patchy vegetation, which would help explain the regression tree output indicating smaller stand-replacing patches on upper slopes.

The association of mid-slope and flatter positions with larger stand-replacing patches may be a product of the topographic and vegetation effects on fire intensity and fire severity, respectively. Mid-slope positions are potentially subjected to higher fire intensity due to the pre-heating of fuels and higher effective wind speeds generally experienced on slopes (Albini 1976; Beaty and Taylor 2008). Elevated fire intensities, as explained previously, increase the potential for high-severity fire effects, and may explain the larger stand-replacing patches. The finding indicating larger patches on flatter positions may reflect the distribution of vegetation rather than an actual topographic effect. Recall that the split in the regression tree based on topographic position occurred within the group of vegetation that included shrubland, lodgepole pine, and Jeffrey pine (Fig. 5). Of these dominant vegetation types, lodgepole pine accounts for over 70% of the area classified as flat topographic position. Although lodgepole pine stands in the Sierra Nevada are not associated with the extensive crowning that lodgepole pine stands in the Rocky Mountains experience (Romme 1982), Caprio (2006, 2008) demonstrate that moderate-sized patches of stand-replacing fire have occurred historically in Sierran lodgepole stands. An alternative explanation to both of these hypotheses is that since the mid slope position and flat classifications account for such a large proportion of the total Illilouette study area (68%), our results could simply reflect the larger range of possible patch sizes rather than a true topographic effect. It is possible that either the 500 m focal mean we used to calculate topographic position or the topographic position index thresholds we used (see Jenness 2007) were not optimal. Perhaps subsequent studies can explore the effect of varying both focal mean sizes and topographic position index thresholds on characterizing spatial fire patterns.

The finding that within pine-dominated and shrubland vegetation types smaller patches tended to occur in areas that burned more recently (10–16 yr), while larger stand-replacing patches tended to occur in older burns (17–30 yr) is consistent with previous work indicating lower fire severity where intervals between fires were shorter (Collins et al. 2007). The fact that patches in which no previous fires were detected during the WFU period (labeled >30 yr in Fig. 5) also tended to be smaller seems counterintuitive. One might think that areas where fire was absent the longest would tend to have larger patches due to the accumulation of surface fuels and increases in tree density (Parsons and Debenedetti 1979). However, based on our field observations (B. Collins, unpublished data) it was apparent that many pine-dominated areas that had not burned in the WFU period contained highly discontinuous fuel beds and/or were adjacent to natural fire breaks (e.g., large rock outcrops). This discontinuity in fuels could be attributed to overall low site productivity or presence of rock within the stands. The patchy surface fuels and presence of natural fire breaks in these previously unburned areas may explain the smaller stand-replacing patches, particularly in the Jeffrey pine-dominated areas.

Based on the relatively low fit of the regression tree using stand-replacing patch sizes as the dependent variable, it is apparent that other factors beyond those captured in our spatial analysis are driving observed variability in patch sizes. We assembled fairly detailed datasets of predictor variables, which previous studies have identified as influencing fire severity, e.g.,: (1) fire weather, applied in a spatially explicit manner using fire progression maps (Collins et al. 2007; Roman-Cuesta et al. 2009), (2) topographic position (Beaty and Taylor 2008), and (3) time since previous fire (Collins et al. 2007, 2009). Despite this effort, our statistical model only explained 26% of the variance in stand-replacing patch sizes. This suggests that more local factors of fuel load, vegetation structure, and weather are probably driving stand-replacing patch sizes. The wide range in stand-replacing patch sizes among the Hoover and Meadow fires (Fig. 4), combined with this lack of explanatory power demonstrates the complexity of mixed-severity fire in these forests. Obtaining more spatially explicit estimates of surface fuels, vegetation structure, and weather would be extremely difficult at landscape scales. Improved techniques of processing LiDAR imagery could help with quantifying vegetation structure, but characterizing fine-scale surface fuels and local weather still pose a significant challenge.

Our analysis of potential vegetation constraint on stand-replacing patch sizes revealed an important distinction between shrub- and conifer-dominated vegetation types. Stand-replacing patches tended to be more constrained to the underlying vegetation patch when they occurred in shrub-dominated areas, while conifer-dominated stand-replacing patches demonstrated much less evidence of vegetation constraint (Fig. 8). The high proportion of variance explained by this split in the regression tree, and the high overall model fit, leads us to infer that in shrub-dominated areas the extent of high-severity burning is largely driven by vegetation itself (Nagel and Taylor 2005), and that topography and weather play lesser roles. The same inference cannot be made with regard to conifer-dominated stand-replacing patches. While the regression tree split within conifer types indicates lodgepole pine- and Jeffrey pine/shrub-dominated stand-replacing patches had some evidence of vegetation constraint, the relatively high average BufferVeg suggests other factors in addition to the extent of the vegetation patch limited the extent of high-severity burning (Fig. 8). The high BufferVeg values associated with stand-replacing patches that occurred in fir-dominated or co-dominated stands indicates that the extent of the vegetation patches in these types had little or no effect on the extent of the stand-replacing patch (Fig. 8). This is not to contradict our previous assertion that fir-dominated stands may be more susceptible to crown-fire and larger stand-replacing patches, but rather make the point that the other factors (weather, local fuels and topography, etc.) contributed to the actual extent of the stand-replacing patch rather than simply the boundary of a dominant vegetation patch. Steeper slopes further decreased any evidence vegetation constraint, likely due to increased fire intensity, as explained previously.

Summary and management implications

Our characterization of not only the distribution of stand-replacing patch sizes, but of the processes driving the variability in patch sizes can provide insight into vegetation–fire dynamics of upper-elevation mixed-conifer forests in the Sierra Nevada. While high-severity fire represents a fairly low proportion of the total burned area (~15%) stand-replacing patches should be considered an important component shaping these forests. Given the 30+ years of allowing natural fires to burn within the Illilouette basin, our results suggest that the natural range of variability in these forests consists of many small stand-replacing patches (<4 ha) and few large patches (>60 ha).

Using data on fuels, weather, and topography within and around stand-replacing patches we demonstrate that these patches tend to be smaller in shrub-dominated and pine-dominated vegetation types, and larger in more shade-tolerant, fir-dominated types. Additionally, in shrub-dominated types stand-replacing patches are often constrained to the underlying patch of vegetation, which were often smaller than vegetation patches for the other dominant vegetation types we analyzed. In white and red fir types we found little evidence for vegetation patch constraint on the extent of stand-replacing patches. We also demonstrate that larger stand-replacing patches are associated with more severe fire weather, as well as mid-slope and flatter topographic positions. While weather and topography partially explain variability in stand-replacing patch sizes our analysis suggests that fuels (dominant vegetation type) have a higher relative importance in explaining the stand-replacing patch dynamics in these upper elevation mixed-conifer forests. Note that this is under the current, fire regime conditions resulting from the long-established WFU program in the Illilouette basin. More detailed weather data would be needed to more comprehensively explore this relationship.

The patch dynamics we identified can be used to inform management strategies for fire-excluded landscapes in similar forest types. Our results suggest that such efforts may include creating stand-replacing patches across a low, but significant, portion of the landscape. This patch creation could mimic a more natural distribution of patch sizes across a landscape (Bergeron et al. 2007). For landscapes in which restoring this type of pattern through fire alone is unattainable due to management and societal constraints (e.g., sensitive habitat, human communities, smoke production, etc.), even-aged silviculture with inclusion of legacy structures (Franklin et al. 2002) may be appropriate on a relatively small proportion of the landscape. In these more constrained landscapes even-aged approaches allow for a much greater degree of control on where patches can and cannot be created, as compared to allowing fires to 'run their course.' Ultimately silvicultural prescriptions that attempt to mimic natural forest–fire dynamics may never achieve the complexity that freely burning fire can. Expanding the domain for allowing more freely burning wildland fire within upper elevation mixed-conifer forests will increase patch heterogeneity across landscapes and likely increase forest resiliency.

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References

Agee JK (1998) The landscape ecology of Western forest fire regimes. Northwest Sci 72:24–34

Albini FA (1976) Estimating wildfire behavior and effects. General Technical Report INT-30. U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station, Ogden, UT, p 92

Beaty RM, Taylor AH (2008) Fire history and the structure and dynamics of a mixed conifer forest landscape in the northern Sierra Nevada, Lake Tahoe Basin, California, USA. For Ecol Manag 255:707–719

Bergeron Y, Drapeau P, Gauthier S, Lecomte N (2007) Using knowledge of natural disturbances to support sustainable forest management in the northern Clay Belt. Forest Chron 83:326–337

Bradshaw LS, Deeming JE, Burgan RE and Cohen JD (1984) The 1978 national fire danger rating system: technical documentation. General Technical Report INT-169. U. S. Department of Agriculture, Forest Service, Intermountain and Range Experiment Station, Ogden, UT, p 44

Breiman L, Friedman JH, Olshen RA, Stone CG (1984) Classification and regression trees. Wadsworth, Belmont, CA, USA

Brown PM, Kaufmann MR, Shepperd WD (1999) Long-term, landscape patterns of past fire events in a montane ponderosa pine forest of central Colorado. Landscape Ecol 14:513–532

Brown PM, Wienk CL, Symstad AJ (2008) Fire and forest history at Mount Rushmore. Ecol Appl 18:1984–1999

Caprio AC (2006) Fire history of lodgepole pine in the southern Sierra Nevada, California. In: 3rd international fire ecology and management congress changing fire regimes: context and consequences. Association for Fire Ecology, San Diego, California, pp 1–5

Caprio AC (2008) Reconstructing fire history of lodgepole pine on Chagoopa Plateau, Sequoia National Park, California. General Technical Report PSW-GTR-189. U. S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, p 7

Collins BM, Stephens SL (2007) Managing natural wildfires in Sierra Nevada wilderness areas. Front Ecol Environ 5:523–527

Collins BM, Kelly M, van Wagtendonk JW, Stephens SL (2007) Spatial patterns of large natural fires in Sierra Nevada wilderness area. Landscape Ecol 22:545–557

Collins BM, Miller JD, Thode AE, Kelly M, van Wagtendonk JW, Stephens SL (2009) Interactions among wildland fires in a long-established Sierra Nevada natural fire area. Ecosystems 12:114–128

De'ath G (2002) Multivariate regression trees: a new technique for modeling species–environment relationships. Ecology 83:1105–1117

De'ath G, Fabricius KE (2000) Classification and regression trees: a powerful yet simple technique for ecological data analysis. Ecology 81:3178–3192

Franklin JF, Spies TA, Van Pelt R, Carey AB, Thornburgh DA, Berg DR, Lindenmayer DB, Harmon ME, Keeton WS, Shaw DC, Bible K, Chen JQ (2002) Disturbances and structural development of natural forest ecosystems with silvicultural implications, using Douglas-fir forests as an example. Forest Ecol Manag 155:399–423

Girvetz EH, Greco SE (2007) How to define a patch: a spatial model for hierarchically delineating organism-specific habitat patches. Landscape Ecol 22:1131–1142

Hessburg PF, Agee JK, Franklin JF (2005) Dry forests and wildland fires of the inland Northwest USA: contrasting the landscape ecology of the pre-settlement and modem eras. Forest Ecol Manag 211:117–139

Hessburg PF, Salter RB, James KM (2007) Re-examining fire severity relations in pre-management era mixed conifer forests: inferences from landscape patterns of forest structure. Landscape Ecol 22:5–24

Holden ZA, Morgan P, Crimmins MA, Steinhorst RK, Smith AM (2007) Fire season precipitation variability influences fire extent and severity in a large southwestern wilderness area, United States. Geophys Res Lett 34:L16708

Jenness J (2007) Some thoughts on analyzing topographic habitat characteristics. Jenness Enterprises, Flagstaff, AZ, USA, p 26

Miller JD, Thode AE (2007) Quantifying burn severity in a heterogeneous landscape with a relative version of the delta Normalized Burn Ratio (dNBR). Remote Sens Environ 109:66–80

Miller JD, Safford HD, Crimmins M, Thode AE (2009a) Quantitative evidence for increasing forest fire severity in the Sierra Nevada and southern Cascade Mountains, California and Nevada, USA. Ecosystems 12:16–32

Miller JD, Knapp EE, Key CH, Skinner CN, Isbell CJ, Creasy RM, Sherlock JW (2009b) Calibration and validation of the relative differenced Normalized Burn Ratio (RdNBR) to three measures of fire severity in the Sierra Nevada and Klamath Mountains, California, USA. Remote Sens Environ 113:645–646

Nagel TA, Taylor AH (2005) Fire and persistence of montane chaparral in mixed conifer forest landscapes in the northern Sierra Nevada, Lake Tahoe Basin, California, USA. J Torrey Bot Soc 132:442–457

North M, Stine PA, O'hara KL, Zielinski WJ and Stephens SL (2009) An ecosystems management strategy for Sierra mixed-conifer forests. General Technical Report PSW-GTR-220. U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, CA, p 49

Ott RL, Longnecker M (2001) An introduction to statistical methods and data analysis, 5th edn. Duxbury, Thomson Learning, Pacific Grove

Parsons DJ, Debenedetti SH (1979) Impact of fire suppression on a mixed-conifer forest. Forest Ecol Manag 2:21–33

Pyne SJ, Andrews PL, Laven RD (1996) Introduction to wildland fire, 2nd edn. John Wiley & Sons, Inc., New York

Rollins MG, Morgan P, Swetnam T (2002) Landscape-scale controls over 20th century fire occurrence in two large Rocky Mountain (USA) wilderness areas. Landscape Ecol 17:539–557

Roman-Cuesta RM, Gracia M, Retana J (2009) Factors influencing the formation of unburned forest islands within the perimeter of a large forest fire. Forest Ecol Manag 258:71–80

Romme WH (1982) Fire and landscape diversity in subalpine forests of Yellowstone National Park. Ecol Monogr 52:199–221

Safford HD, Miller JD, Schmidt D, Roath B, Parsons A (2008) BAER soil burn severity maps do not measure fire effects to vegetation: a comment on Odion and Hanson (2006). Ecosystems 11:1–11

Schwilk DW, Keeley JE, Bond WJ (1997) The intermediate disturbance hypothesis does not explain fire and diversity pattern in fynbos. Plant Ecol 132:77–84

Scott JH and Reinhardt ED (2001) Assessing crown fire potential by linking models of surface and crown fire behavior. Research Paper RMRS-RP-29. U. S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, p 59

Stephens SL, Fry DL (2005) Spatial distribution of regeneration patches in an old-growth *Pinus jeffreyi*-mixed conifer forest in northwestern Mexico. J Veg Sci 16:693–702

Stephens SL, Gill SJ (2005) Forest structure and mortality in an old-growth Jeffrey pine-mixed conifer forest in northwestern Mexico. Forest Ecol Manag 205:15–28

Stephens SL, Fry D, Franco-Vizcaíno E (2008) Wildfire and forests in northwestern Mexico: the United States wishes it had similar fire problems. Ecol Soc 13:10

Taylor AH, Skinner CN (1998) Fire history and landscape dynamics in a late-successional reserve, Klamath Mountains, California, USA. Forest Ecol Manag 111:285–301

Taylor AH, Skinner CN (2003) Spatial patterns and controls on historical fire regimes and forest structure in the Klamath Mountains. Ecol Appl 13:704–719

Thompson JR, Spies TA, Ganio LM (2007) Reburn severity in managed and unmanaged vegetation in a large wildfire. Proc Natl Acad Sci USA 104:10743–10748

Turner MG, Romme WH (1994) Landscape dynamics in crown fire ecosystems. Landscape Ecol 9:59–77

Turner MG, Hargrove WW, Gardner RH, Romme WH (1994) Effects of fire on landscape heterogeneity in Yellowstone National Park, Wyoming. J Veg Sci 5:731–742

van Wagtendonk JW (2007) The history and evolution of wildland fire use. Fire Ecol 3:3–17

van Wagtendonk JW, Fites-Kaufman JA (2006) Sierra Nevada bioregion. In: Sugihara NG, van Wagtendonk JW, Shaffer KE, Fites-Kaufman JA, Thode AE (eds) Fire in California's ecosystems. University of California Press, Berkeley, California, USA, pp 264–294

van Wagtendonk JW, Lutz JA (2007) Fire regime attributes of wildland fires in Yosemite National Park, USA. Fire Ecol 3:34–52