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RESEARCH ARTICLE

Spatial patterns of large natural fires in Sierra Nevada wilderness areas

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Abstract The effects of fire on vegetation vary based on the properties and amount of existing biomass (or fuel) in a forest stand, weather conditions, and topography. Identifying controls over the spatial patterning of fire-induced vegetation change, or fire severity, is critical in understanding fire as a landscape scale process. We use gridded estimates of fire severity, derived from Landsat ETM+ imagery, to identify the biotic and abiotic factors contributing to the observed spatial patterns of fire severity in two large natural fires. Regression tree analysis indicates the importance of weather, topography, and vegetation variables in explaining fire severity patterns between the two fires. Relative humidity explained the highest proportion of total sum of squares throughout the Hoover fire (Yosemite National Park, 2001). The lowest fire severity corresponded with increased relative humidity. For the Williams fire (Sequoia/Kings Canyon National Parks, 2003) dominant vegetation type explains the highest proportion of sum of squares. Dominant vegetation was also important in determining fire severity throughout the Hoover fire. In both fires, forest stands that were dominated by lodgepole pine (*Pinus contorta*) burned at highest severity, while red fir (*Abies magnifica*) stands corresponded with the lowest fire severities. There was evidence in both fires that lower wind speed corresponded with higher fire severity, although the highest fire severity in the Williams fire occurred during increased wind speed. Additionally, in the vegetation types that were associated with lower severity, burn severity was lowest when the time since last fire was fewer than 11 and 17 years for the Williams and Hoover fires, respectively. Based on the factors and patterns identified, managers can anticipate the effects of management ignited and naturally ignited fires at the forest stand and the landscape levels.

Keywords Fire ecology · Normalized Burn Ratio · dNBR · Prescribed natural fire · Regression tree · Wildland fire use

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Introduction

Fire is among the most influential processes shaping the structure and pattern of forests in the western U.S. (Romme 1982; Agee 1998; Brown et al. 1999). Fire directly alters vegetation composition and density at the scale of individual forest stands. At the landscape scale, fire affects the size, arrangement, and age structure of vegetation patches (Turner et al. 1994). Fire suppression policies and the removal of Native American ignitions have disrupted these natural interactions between process and structure/pattern. As a result, many of the forested landscapes in the western U.S. have become increasingly homogenized (Agee 1998; Fule et al. 2004; Hessburg et al. 2005).

The Leopold Report (1963) recognized the importance of natural processes in maintaining forested ecosystems, and brought national attention to the unintended landscape changes caused by fire suppression. Following this report, the Wilderness Act (1964) was passed, charging managers of wilderness areas to "preserve natural conditions." In order to meet management objectives aimed at restoring natural conditions, structural characteristics, as well as the processes that govern them, must be restored (Parsons et al. 1986). In response to the Leopold Report and the Wilderness Act, the National Park Service adopted a new policy in 1968 that included fire as management tool to restore forested landscapes (Stephens and Ruth 2005). This policy called for the use of both prescribed and natural fire to achieve restoration objectives.

Due to administrative complexity and the inherent difficulties in allowing natural fire to operate on the landscape, natural fire programs (now referred to as wildland fire use, WFU) have only been implemented in a few discrete locations in the western U.S. (Christensen 1991). Illilouette Creek and Sugarloaf Creek basins in Yosemite National Park and Sequoia/Kings Canyon National Parks, respectively, are two such places. For over 30 years managers in both parks have allowed lightning-ignited fires to burn in these isolated wilderness locations. As a result of repeated burning over the last 30 years, Illilouette Creek and Sugarloaf Creek basins, along with a few wilderness areas in the Rocky Mountains (e.g. Rollins et al. 2002), have as close to natural fire regimes as any place in the western U.S.; despite having over half a century of fire suppression prior to the induction of these natural fire programs (Caprio and Graber 2000).

Illilouette Creek and Sugarloaf Creek basins provide a unique opportunity to study the process and effects of fire under relatively natural conditions. Historical studies of pre-Euroamerican fire occurrences have robustly characterized the temporal patterns of natural fires (Romme 1982; Swetnam 1993; Stephens and Collins 2004). Far fewer studies have described the spatial patterns of natural fires (but see Brown et al. 1999; Heyerdahl et al. 2001). In the vegetation types that comprise Illilouette Creek and Sugarloaf Creek basins, it is believed that fires historically burned with highly variable intensities (Agee 1998). High vegetation mortality would occur in patches where fire intensity was elevated; while other areas within the same fire perimeter burning under moderate or low intensities would experience only understory vegetation mortality. This variable fire-induced vegetation mortality, or fire severity, in combination with unburned islands within the fire perimeter, creates a mosaic landscape consisting of distinct forest type and age class patches (Miller and Urban 1999; Fule et al. 2003). Our use of the term fire severity refers to the extent of fire-induced change in dominant vegetation.

While the process of fire influences the composition and pattern of forests over a landscape, the pattern and composition of vegetation influences fire spread and severity (Stephens 2001; Li and Wu 2004). The amount and continuity of live and dead vegetation, or fuel, limits fire spread and severity (Minnich et al. 2000; Rollins et al. 2002). The amount and continuity of fuel depends on the species composition and developmental stage of a forest stand. Immature stands have much lower fuel loads (van Wagtendonk et al. 1998), and as a result do not readily burn. Mature lodgepole pine (*Pinus contorta*) forests, tend to have moderate woody surface fuel loads and high canopy fuel loads (van Wagtendonk et al. 1998; Reinhardt et al. 2000), while mature Jeffrey pine (*Pinus jeffreyi*) forests have low canopy fuel loads and low surface fuel loads that are highly patchy (Stephens 2004, Stephens and Gill 2005). These differences lead to differential heat output under combustion, which generally results in higher expected mortality from a fire burning in a lodgepole pine forest as compared to a fire in a Jeffrey pine forest. In addition to the amount of fuel, moisture content of fuels also influences the combustibility, and consequently the heat output (Rothermel 1972). Multi-strata forest stands with closed tree canopies, such as those seen in mature red fir (*Abies magnifica*) and white fir (*A. concolor*) forests, tend to have higher moisture content in the surface and ground fuels because less light and wind actually reach the forest floor that would desiccate fuels.

Weather and topography also affect the pattern of fire severity over a landscape. Wind supplies oxygen to a fire and increases the rate of combustion (Rothermel 1972). In addition, wind, along with temperature and relative humidity, influences the moisture content of fuels. Higher wind speed, higher temperature, and lower relative humidity, will desiccate fuels, which increases the amount of fuel consumed, thus increasing the fire intensity. Similarly, aspect and slope influence fire intensity due to the differential effects of solar radiation, and fuel preheating, respectively, on fuel moisture content (Pyne et al. 1996).

The objective of this paper is to identify the abiotic and biotic factors responsible for the differential fire effects across the landscape. Very little is known on the controls over spatial patterning of fire severity at the landscape scale. We use satellite imagery and geospatial analysis to study fire severity of two natural wildfires; one occurred in Illilouette Creek basin and the other in Sugarloaf Creek basin. These fires provide recent examples of fire-caused change over large areas composed of several different vegetation types. Investigating the effects of these fires, and the factors driving these effects, is necessary to advance our understanding of how fire shapes landscapes (Finney et al. 2005). Due to the natural fire programs that have been in effect in and around these two natural wildfires, the results from this study can serve as a proxy for understanding the historical range of variability for fire in these ecosystems. This would provide managers a baseline reference for defining restoration goals. We also intend for this analysis to provide managers information that will assist in the implementation of both WFU and prescribed fire programs. Based on the factors and patterns identified, managers can anticipate the effects of management ignited and naturally ignited fires on forest stands, as well as the resulting pattern over the landscape.

Methods

Study area

Yosemite National Park and Sequoia/Kings Canyon National Parks are located in the central and south-central Sierra Nevada, respectively (Fig. 1). Each park is over 300,000 ha and extends from the foothills (~500 m elevation) to the crest of the Sierra Nevada (over 4000 m elevation). The climate is Mediterranean with cool, moist winters, and warm, generally dry summers. Precipitation varies with elevation and is predominantly snow, with annual averages near 100 cm. (Caprio and Graber 2000; van Wagtendonk et al. 2004).

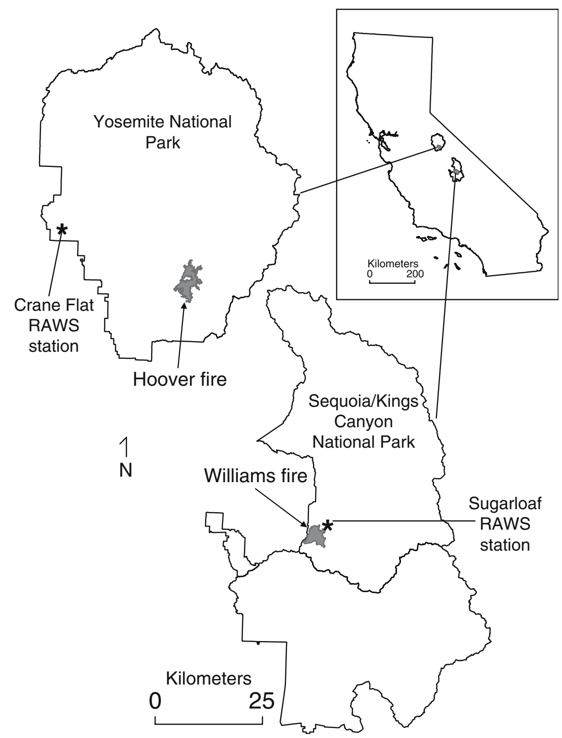

Fig. 1 Locations of the Hoover (Yosemite NP) and Williams (Sequoia/Kings NP) fires, California, USA

Vegetation in both parks also varies with elevation. Oak woodlands and chaparral shrublands dominate lower elevations, with mixed conifer forests dominating the mid-elevations, and subalpine forests in the high elevation (see for detailed explanations of vegetation: Caprio and Graber 2000; van Wagtendonk et al. 2004). The dominant forest types found in Illilouette Creek and Sugarloaf Creek basins are Jeffrey pine (*Pinus jeffreyi*), lodgepole pine (*Pinus contorta*), white fir (*Abies concolor*), red fir (*Abies magnifica*), and are interspersed with meadows and shrublands.

The Hoover fire (started July 26, 2001) and the Williams fire (started August 8, 2003) were both lightning-ignited fires that were allowed to burn unsuppressed as part of the WFU programs in Illilouette Creek and Sugarloaf Creek basins, respectively. These fires were selected because they were recent, relatively large fires that burned in areas with established WFU programs. The Hoover fire burned over 2100 ha and the Williams fire burned nearly 1,400 ha. Table 1 shows the total area burned in each fire by dominant vegetation type. The weather conditions during the time these fires burned, as well as the topography within the fire perimeters, are summarized in Table 2.

Spatial data

Fire severity data for both fires was assessed using the differenced Normalized Burn Ratio (dNBR), which was obtained from the National Park Service–U.S Geological Survey Burn Severity Mapping Project (http://burnseverity.cr.usgs.gov/fire_main.asp). This index is derived by differencing reflectance in bands 4 and 7 in pre- and post-fire scenes from Landsat ETM+ imagery. The dNBR is susceptible false identification of fire-induced vegetation change, particularly with respect to clouds in LANDSAT scenes, as well as seasonal differences plant moisture content and plant phenology. Key and Benson (2005) control for these potential problems by selecting cloud free scenes and by obtaining images from similar seasonal periods. This provides a flexible and robust method for characterizing fire severity (Brewer et al. 2005). Ground-based validation of fire severity showed strong correlation with dNBR values throughout the extent of the Hoover fire (van Wagtendonk et al. 2004). These dNBR images are in raster format, at 30 m spatial resolution, and import directly into a Geographic Information System (GIS). Figure 2 shows the images for the two fires classified by fire severity rating using the range of values recommended by Key and Benson (2005). The dNBR is a continuous variable that ranges from –550 to 1350 (Key and Benson 2005).

For each of the pixels (30 m) in both dNBR images we assigned values for vegetation type, weather, topography, and previous fire history. The vegetation type values were based on vegetation maps provided by each National Park. Due to differences in classification schemes, vegetation in each park was re-classified in the eight categories listed in Table 1. We use dominant vegetation type under the assumption that fuel amounts and fuel structure corresponded with vegetation type.

Table 1 Area burned in Hoover and Williams fires summarized by dominant vegetation type

	Hoover Fire (2001) Yosemite National Park			Williams Fire (2003) Sequoia/Kings Canyon National Park		
	Number of cells (30 × 30 m)	Hectares	%	Number of cells (30 × 30 m)	Hectares	%
Abies concolor	63	5.7	< 1	5009	450.8	32
Abies magnifica	11685	1051.7	49	6303	567.3	41
Juniperus occidentalis	103	9.3	< 1	–	–	–
Pinus contorta	3961	356.5	17	1925	173.3	12
Pinus jeffreyi	5885	529.7	25	1646	148.1	11
Meadow	674	60.7	2	280	25.2	2
Shrubland	1101	99.1	5	102	9.2	1
Bare rock/water	200	18	1	110	9.9	1
Totals	23672	2131		15375	1384	

Table 2 Summary statistics for the weather and topographic variables used in the regression tree analysis of fire severity throughout the Hoover and Williams fires

	Hoover Fire (July–October, 2001) Yosemite National Park					Williams Fire (August–November, 2003) Sequoia/Kings Canyon National Park				
	Temperature (°C)	Relative humidity (%)	Wind gust speed (m/s)	Slope (%)	Elevation (m)	Temperature (°C)	Relative Humidity (%)	Wind gust speed (m/s)	Slope (%)	Elevation (m)
Mean	23.9	23.3	7.6	19.4	2340	22.0	17.5	5.0	21.7	2542
Median	25.1	21.2	7.9	16.3	2340	21.6	12.2	4.8	19.4	2528
Range	18.8–27.5	17.5–39.8	5.4–9.2	0–79.9	2108–2681	1.8–26.0	6.9–65.5	2.8–8.5	0–71.7	2275–2898
Standard deviation	2.9	5.7	1.1	13.4	101	3.1	10.8	1.2	12.1	124

The weather variables for the Hoover fire were obtained from the Crane Flat Lookout Remote Automated Weather Station (RAWS), and from the Sugarloaf RAWS for the Williams fire, which were the stations nearest to each fire that had complete data sets for entire burning period of each fire (see Fig. 1). We averaged hourly values of temperature, wind speed gusts, and relative humidity to get daytime (10 am–5 pm) estimates of each variable. These daytime estimates were averaged again over the number of days included in each burning period represented on the fire progression maps (Fig. 3). These fire progression maps were produced by the fire management staff at each park throughout the duration of the two fires. The progression maps include daily fire perimeters during highly active burning periods, and up to several days or weeks during less active burning periods. The averaged weather variables for a given burning period were assigned to every pixel within that perimeter. This relatively coarse application of weather variables may be tenuous, especially when the burning period exceeds several days. However, the burning periods that do exceed several days appear to affect a lower proportion of the area in each fire, based on the fire progression maps (Fig. 3). We assume that averaging over all the days included in a given burning period captures the general conditions. We feel this method is the best way to incorporate actual weather data into an analysis explaining observed fire severity.

The topographic variables, aspect (degrees) and slope (percent), were derived from 30 m digital elevation models (DEM) (obtained from the GIS specialist at each park). The DEMs were clipped using the perimeter of each fire to obtain only those pixels affected by each fire. Due to aspect being circular variable (0 and 360 are the same) we used a sine transformation to maintain east-west orientation and a cosine transformation to maintain north-south orientations. Previous fire history was assessed using digitized fire atlases, which included all fires that occurred in both the Illilouette and Sugarloaf basins since 1972 (e.g. Rollins et al. 2001). Based on overlapping fire perimeters we created a previous burn frequency variable, ranging from 0 to 4 times. In addition, we used the digital fire

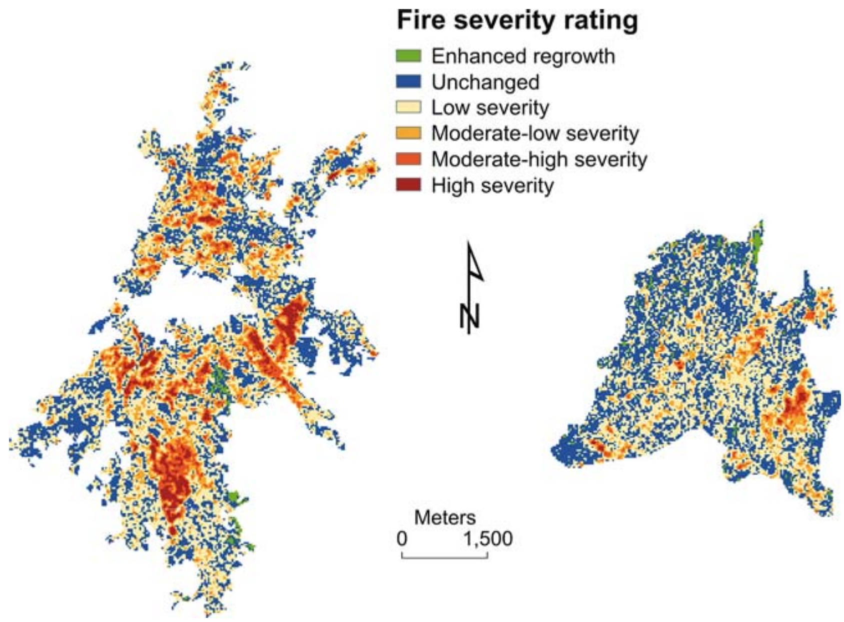

Fig. 2 The differenced Normalized Burn Ratio images for the Hoover (**Left**) and Williams (**Right**) fires classified into fire severity ratings

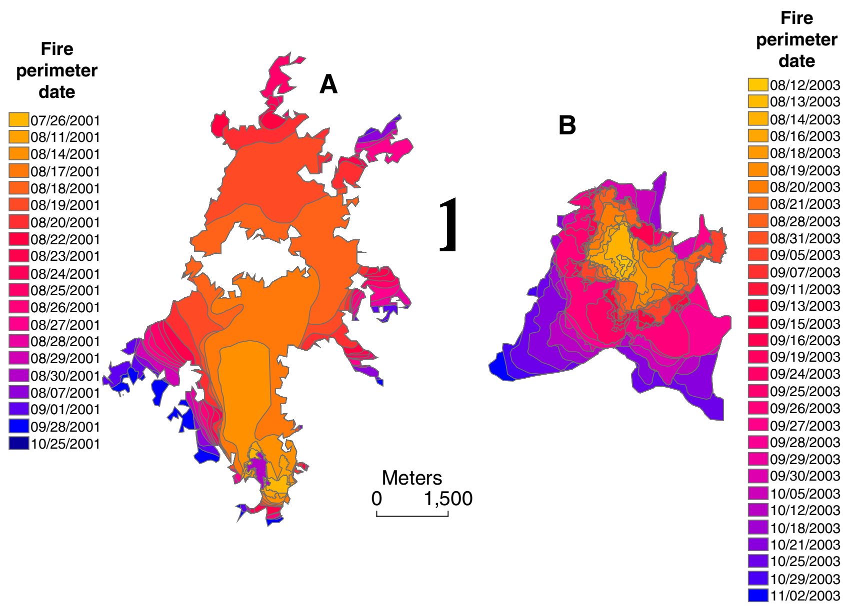

Fig. 3 Progression maps for the Hoover (**A**) and Williams (**B**) fires. Weather variables were applied to pixels based on these burning period perimeters

atlases to create a time since last fire variable based on the perimeter of the most recent fire. In areas that have no record of fire from 1972 on we assigned a value of 40 years. Each 30 m pixel within the dNBR images was assigned a value for each of the variables mentioned (vegetation type, temperature, relative humidity, wind speed, slope, aspect, previous burn frequency, and time since last fire) ending up with a total of 23,672 pixels for the Hoover fire and 15,375 for the Williams fire. We use FRAGSTATS to compute the area-weighted mean patch sizes for each fire severity class (McGarigal et al. 2002). Area-weighted means place more emphasis on larger patches and less on the numerous smaller patches (one to four 30 m cells) that account for over half the total number of patches in each severity class.

Statistical analyses

We explored possible relationships between each of the independent variables mentioned previously and the response variable, dNBR, 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, regression trees convey relationships clearly, which allows for easy interpretation of the results. The regression tree is constructed by repeatedly splitting the data into increasingly homogenous groups based on the response variable. Each split minimizes the sum of squares within the resulting groups. The number of terminal nodes, or leaves, 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 leaves.

One potential problem with using regression tree analysis on spatial data is the lack of independence among observations. Semivariogram analysis on the dNBR images indicated spatial autocorrelation in fire severity estimates up to 1,000 m for the Hoover fire and 750 m for the Williams fire. Ideally, one would choose to perform the analysis on a subset of data that are separated beyond the distance of autocorrelation. However, in this study such an approach would only allow for a total number of 24 observations for the Hoover fire and 14 for the Williams fire. Calbk et al. (2002) examined the ability of regression tree analysis to handle spatial autocorrelation and found that regression trees were ''able to effectively model correlative relationships despite autocorrelation in the original data.'' Based on the fact that Calbk et al. (2002) used data that were structured similar to ours (raster based) we submit that regression tree analysis is appropriate for this study.

Results

The frequency distributions for the dNBR images show that the Hoover fire burned with a greater proportion of moderate and high severity, while the Williams fire had a higher proportion of unchanged area within the fire perimeter (Fig. 4). The maps of fire severity (Fig. 2) illustrate this, showing several large patches of higher severity throughout the Hoover fire, compared to only a few isolated high severity patches in the Williams fire. In addition, the area-weighted mean patch sizes for each fire-severity rating class (Fig. 4) show that high and moderate-high severity patches are larger throughout the Hoover fire, while unchanged and low-severity patches were much larger throughout the Williams fire. Despite these differences, an overwhelming majority of area within both fire perimeters is in the unchanged and low severity classes (Figs. 2, 4).

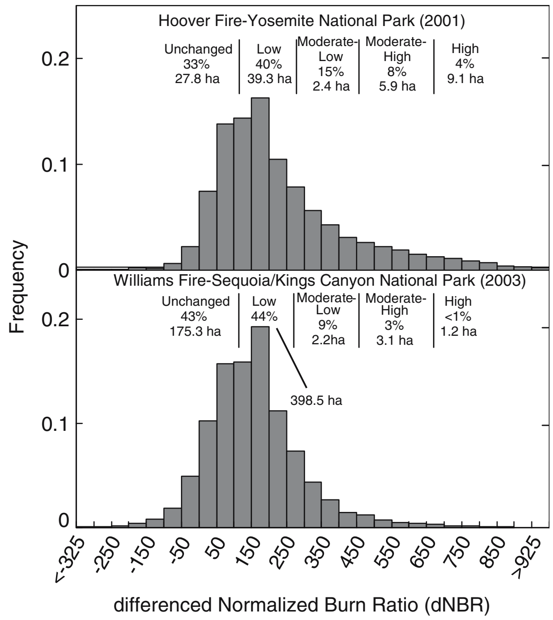

Fig. 4 Frequency distributions for differenced Normalized Burn Ratio values for the Hoover and Williams fires, along with the proportion of pixels and area-weighted mean patch size by fire severity rating class for each fire. The fire severity classes are based on Key and Benson (2005)

The regression tree analysis indicates differences in the relative importance of weather, topography, and vegetation in explaining fire severity patterns between the two fires. Relative humidity explained the highest proportion of total sum of squares (SS) throughout the Hoover fire (Fig. 5). The lowest dNBR values corresponded

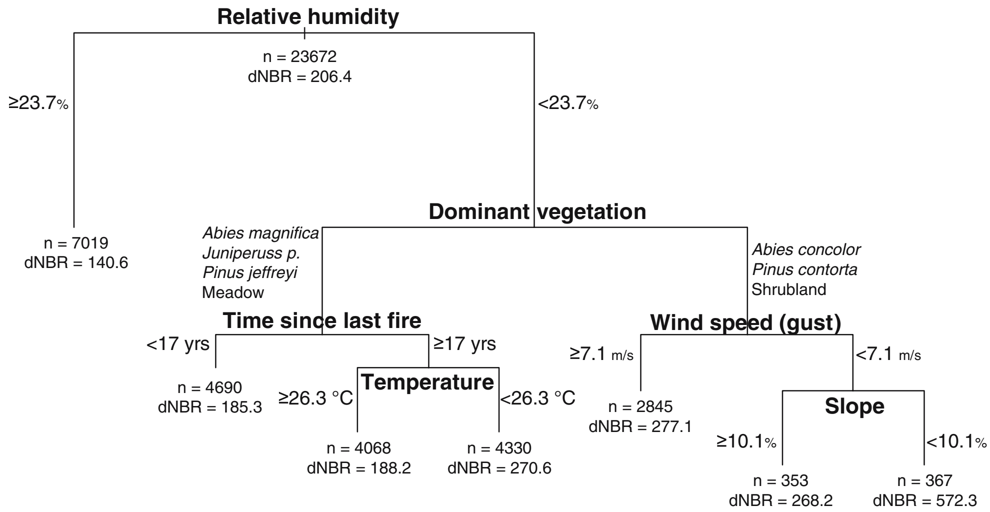

Fig. 5 Regression tree explaining the spatial distribution of differenced Normalized Burn Ratio (dNBR) values throughout the Hoover fire. The number of pixels, along with the average dNBR, in the resulting group is reported at each node. The length of the line from each split indicates the relative proportion of total sum of squares explained by that split. The total R^2 for the tree is 14%

with increased relative humidity, which by itself best explained the observed pattern of burn severity in the range of unchanged and low severity. This is the reason for the lopsided shape of the Hoover fire regression tree. For the Williams fire, relative humidity does not show up at all on the tree (Fig. 6). Instead, the dominant vegetation type explains the highest proportion of SS, which accounts for a much higher relative proportion of total SS. Dominant vegetation is subsequently split on both sides of the Williams fire regression tree, which improves the total SS explained, and ultimately leads to a more balanced regression tree.

Dominant vegetation was second in terms of its importance in explaining fire severity throughout the Hoover fire. Both fires consistently burned at higher severity in forest stands dominated by lodgepole pine (*Pinus contorta*) and a lower severity in red fir (*A. magnifica*) stands and meadow vegetation. The two fires were inconsistent in that throughout the Williams fire Jeffrey pine (*Pinus jeffreyi*) was associated with higher burn severity, while white fir and shrublands were associated with lower burn severity. The opposite was true for the observed burn severity in each of these dominant vegetation types throughout the Hoover fire. However, it is important to note that white fir only dominated a very small percentage (<1%) of the Hoover fire area (Table 1).

In the vegetation types associated with the lowest dNBR values throughout the Williams fire (white fir, red fir, meadow, and shrubland) higher air temperatures led to increased burn severity. Lower air temperature, in these vegetation types, resulted in a subsequent split in which time since last fire explains the differentiation between areas that burned under the lowest (or within the unchanged range) and second lowest burn severity. Lower time since last fire (<17 years) was similarly associated with low burn severity throughout the Hoover fire. When time since last fire was greater than 17 years, higher dNBR values tended to coincide with lower air temperature. This is opposite of the burn severity—air temperature association for the Williams fire.

In the vegetation types associated with the highest dNBR values, wind speed was the next most important explanatory variable in both fires. However, burn severity in the two fires showed contrasting associations with wind speed (average wind gusts). In the Hoover fire, lower wind speeds corresponded with higher dNBR values, while the opposite was true for the Williams fire (Fig. 6). In the Williams fire, the highest fire severity occurred in Jeffrey pine and lodgepole pine stands when

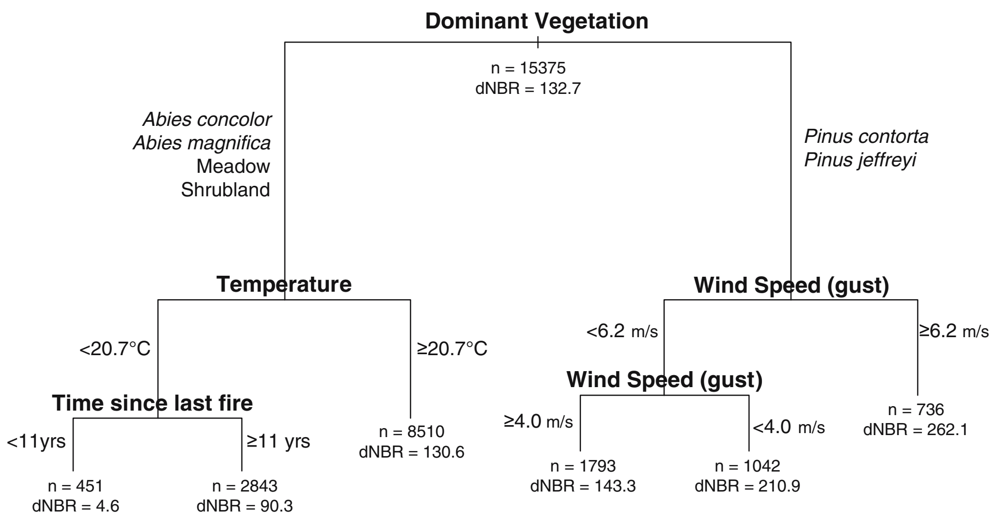

Fig. 6 Regression tree explaining the spatial distribution of differenced Normalized Burn Ratio (dNBR) values throughout the Williams fire. The tree is drawn and labeled as in Fig. 5. The total R^2 for the tree is 12%

average wind gusts exceeded 6.2 m/s. When average wind gusts were under 6.2 m/s the data again split based on wind speed. However, the rule for this split was opposite that of the prior split, indicating the second highest fire severity was associated with the lowest wind speeds, while intermediate fire severity was associated with moderate wind speed.

Aspect and previous burn frequency did not contribute to the explanation of observed dNBR patterns for either fire. Slope did improve to the total proportion of SS in the Hoover fire, indicating the highest fire severity occurred in shrublands, lodgepole pine, and white fir stands, on flatter slopes (< 10.1%), and under lower wind speeds (< 7.1 m/s). Overall, the R^2 for the Hoover and Williams fire regression trees was 14% and 12%, respectively. Relative error was 0.87 for the Hoover fire and 0.89 for the Williams fire.

Discussion

Comparisons between historical (pre-Euro American settlement) fires and the fires studied in the paper are imperfect for a number of reasons: difficulties in reconstructing spatial patterns of historical fires, differences in spatial resolutions, and differences in forest types studied. Given these limitations, the Hoover and Williams fires appear to resemble fires described in the few studies that have attempted to characterize the spatial and structural properties of mixed-severity fires (e.g. Agee 1998; Brown et al. 1999; Fule et al. 2003). The large areas that burned under low-severity, interspersed with unburned patches and patches of high fire-induced mortality created a mosaic pattern across each respective landscape (Fig. 2). Areas that burned under high severity will most likely be regenerated with even-aged cohorts, while areas that experienced moderate and low fire severity will have little or no fire-initiated cohorts (Fule et al. 2003; Schoennagel et al. 2004). With repeated fires interacting on a landscape over time the interspersion of these regeneration pockets among such high proportions of low severity and unchanged area in each fire creates an incredibly complex and heterogeneous landscape. This spatial complexity has been one of the primary obstacles hindering the study of historical fire patterns.

The mean patch sizes (area-weighted) in each fire severity rating class suggest much different landscape fragmentation resulting from the two fires (Fig. 4). Despite the similar relative proportions of each landscape in the unchanged and low-severity classes, the aggregation of area in these classes was quite different. Large contiguous areas within the Williams fire perimeter were minimally altered or unaltered by fire, while higher severity areas tended to occur in small, discrete pockets. This is in contrast to the more fragmented landscape created by the Hoover fire, where mean patch sizes were much smaller in the unchanged and low severity classes and high severity patch sizes were larger.

The high relative importance of dominant vegetation in both regression trees provides insight into the noticeable differences in aggregations of high and low severity pixels in the dNBR images of Hoover and Williams fires (Fig. 2). Both regression trees relate higher fire severity to lodgepole pine stands, while low severity and unchanged areas are associated with red fir stands, and for the Williams fire white fir as well. Over 70% of the area within the Williams fire perimeter is either red or white fir, while for the Hoover fire red fir stands make up approximately 50% of the area (Table 1). In addition, relative abundance of lodgepole pine is higher in the Hoover fire compared to the Williams fire. These differences in relative abundances of fir and lodgepole pine may explain the observed differences in fire severity frequency distributions (Fig. 2) and resulting patches of high and low severity between the two fires (Fig. 4). The discrepancy between the regression trees for the two fires with respect to Jeffrey pine burning under higher or lower severity is difficult to explain. Given the structure of surface and ground fuels we would expect generally lower severity burning in Jeffrey pine forests, as identified in the Hoover fire (Stephens 2004; Stephens and Gill 2005). Perhaps the accuracy of the vegetation data for the Williams fire was not as good, given that it is coarser and much older than that for the Hoover fire.

Previous studies have associated lodgepole pine forests in the central Rockies and in the southern Cascades with infrequent, higher severity fire, which in most cases serves as the dominant regeneration mechanism for lodgepole pine (Romme 1982; Schwilk and Ackerly 2001; Taylor and Solem 2001). This higher fire severity is generally attributed to the greater amounts and continuity of both surface and canopy fuels in mature lodgepole pine stands, which facilitates combustion and leads to higher fire intensity. Red fir and white fir stands, on the other hand, tend to have a more densely packed fuel beds (van Wagtendonk et al. 1998), which slows desiccation and leads to higher moisture content. As a result, fires tend to burn unevenly and generally do not result in extensive high severity patches (Pitcher 1987). The combination of greater abundance of lodgepole pine and lower abundance of red and white fir throughout the Hoover fire most likely explains the larger high severity patches (Fig. 2) and increased area under the high and moderate-high fire severity rating (Fig. 4).

The role of weather in explaining the differences in fire severity distributions between the Hoover and Williams fires is somewhat more ambiguous. Clearly weather influenced fire severity in both fires, as indicated by the regression tree analysis (Figs. 5, 6). Summary statistics for weather indicate generally higher temperatures and wind speeds during the Hoover fire, but lower relative humidity during the Williams fire (Table 2). The lower relative humidity during the Williams fire, along with the absence of relative humidity in the regression tree, suggests that relative humidity may not have been limiting fire behavior. This is in contrast to the Hoover fire, which burned under more moist conditions. Under these conditions, the effect of fluctuating relative humidity on the moisture status of finer fuel particles may have had a more noticeable influence on fire behavior, and thus fire severity. The high explanatory power of relative humidity in the Hoover fire suggests that this split in the regression tree may be identifying a threshold, which when exceeded results in unchanged or very low fire severity. The absence of relative humidity in the Williams fire regression tree may indicate that relative humidity values were mostly below this threshold. However, the lower temperatures during the Williams fire, and the relatively high explanatory power of temperature in the regression tree, may indicate a similar threshold for the Williams fire based on temperature.

In the regression trees for both fires the split immediately following the dominant vegetation group containing lodgepole pine is based on wind (Figs. 5, 6). Based on fire behavior models, and observed fire behavior from actual fires, higher wind speed lead to increased rate of spread, which generally leads to increased fire intensity

(Rothermel 1972). The contradictory nature of the two splits makes it difficult to draw conclusions. In the case of the Hoover fire it appears that fire severity is not necessarily driven by rapid spread rate. Perhaps lower wind speeds correspond with longer residence time of fire. Additionally, lower wind speeds generally correspond with higher scorch height (Andrews et al. 2003), which would tend to lead to higher fire severity. The second split based on wind speed for the Williams fire indicates that the group with the second highest fire severity burned during the lowest wind speeds, which further substantiates the explanation of lower spread rates leading to higher fire severity.

We recognize that topographic and stand structural characteristics influence local wind patterns, as well as temperatures and relative humidities, throughout both fires. As a result, the wind speed, temperature, and relative humidity observations from the weather stations, which in the case of the Hoover fire are from ca. 30 km away, do not necessarily capture the variability in local weather patterns during burning. Additionally, the averaging of the weather variables over multiple days, and subsequent application of these averages over relatively large areas reduces reliability in the relationships identified. However, we feel that because the weather data we used are actual observations, from relatively nearby weather stations they are the best estimates available. Furthermore, the substantial explanatory power that relative humidity, temperature, and wind speed provide in explaining observed dNBR patterns suggest a real connection between the weather estimates and burn severity.

The fact that time since last fire partially explained the observed burn severity in both fires, while previous burn frequency did not for either fire, emphasizes an important distinction. Apparently, what impacts fire severity is not the number of times an area burned previously; rather it is the length of time allowed for fuels to accumulate. Based on the two fires studied, the time it takes for fuels to accumulate to a point at which previous fires no longer impact burn severity in subsequent fires is 11–17 years. This length of time is longer than what Finney et al. (2005) found in studying the impact of prescribed fires on fire severity in a large Arizona wildfire. They found that previous prescribed fires reduced fire severity only if the burns occurred less than four years prior to the wildfire. The extreme weather under which this large Arizona wildfire burned, as well as differences in the vegetation types burned, may account for this apparent discrepancy between Finney et al. (2005) and this study. More work is needed to better understand the temporal extent of fire impacts on burn severity in subsequent fires.

It is important to note that in both the Hoover and Williams fire regression trees, the split involving time since last fire occurs within the forest types that burned under lower fire severity (red fir and Jeffrey pine in the Hoover fire, red fir and white fir in the Williams fire—Figs. 5 and 6). Based on the fact that the lowest dNBR values (in the range of unchanged and low severity classes) for both fires corresponded with these forest types that burned within the last 11–17 years, these results indicate that fire is potentially self-limiting in red fir, and to some extent white fir and Jeffrey pine forest. It does not appear that this is the case for lodgepole pine, at least within the time scale we studied. The potential for a temporal threshold at which previous fires limit subsequent fires carries important implications to mangers and researchers alike. The field of fire management would benefit greatly from further exploration of this hypothesis.

Summary and management implications

The extent and availability of remotely sensed data pertinent to ecological studies is continually expanding. This expansion requires constant innovation in studying complex landscapes and ultimately enhancing our understanding of the natural processes shaping these landscapes (Rollins et al. 2002). The robust characterization of fire severity using dNBR allowed us to identify the factors driving the spatial patterns of fire severity for natural fires in two Sierra Nevada landscapes. Consistencies among the two regression trees suggest that there are some commonalties that could be applied to other areas throughout the Sierra Nevada. Red fir and to some extent white fir stands tended to burn at lower severities. In addition, higher relative humidity, lower temperatures, and lower time since last fire correspond with lower or moderate fire severity. On the other hand, lodgepole pine stands burning under low wind speeds tended to experience the highest mortality. The arrangement and sizes of patches burned at different severities differs between the two fires, which may be partially controlled by the differential dominance in forest types throughout the two fires. Although only two fires are studied, we feel that the methods are straightforward enough, and the analysis is robust enough to be carried out on additional case studies. Additional studies can further elucidate potential weather and/or time since last fire thresholds, which will ultimately enhance our understanding of fire as a natural process.

Ecologists and managers are increasingly recognizing the importance of fire as a natural ecosystem process. In addition, natural fire plays a critical role in shaping landscapes by promoting heterogeneity among vegetation types and age class patches (van Wagtendonk 1995). In the absence of fire throughout much of the Sierra Nevada, and more generally in drier forests throughout the western U.S. as a whole, landscapes have become homogenized (Miller and Urban 2000; Hessburg et al. 2005). The results from this study characterize the spatial properties and factors driving the patterns of more natural fire-induced vegetation change. Many of the landscapes throughout the Sierra Nevada are not in a state at which large-scale fire will mimic the effects associated with more natural fire. As a result, managers might use the results from this study as guidelines for the implementation of mechanical and prescribed fire treatments aimed towards the ultimate goal of allowing the natural process of fire to operate on the landscape. Additionally, wildland fire managers can use these findings to aid in planning for and using wildland fires to manage ecosystems and landscapes. The factors identified can help determine expected change in the landscape pattern of vegetation resulting from allowing wildfires to burn in various conditions.

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