Final Report
Landscape Scale Effects of Wildland Fire Use Programs in Sierra Nevada Wilderness Areas

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PRINCIPAL INVESTIGATORS

Dr. Jan van Wagtendonk
Research Scientist
U.S. Geological Survey, Western Ecological Research Center
Yosemite Field Station
El Portal, California 95318-0700
Phone 209-379-1306; email: jan_van_wagtendonk@usgs.gov

Dr. Scott Stephens
Assistant Professor of Fire Sciences
137 Mulford Hall - # 3114
Department of Environmental Science, Policy, and Management
University of California, Berkeley, CA. 94720-3114
Phone 510-642-7304; email: stephens@nature.berkeley.edu

Dr. Brandon Collins
Post-doctoral researcher
137 Mulford Hall # 3114
Department of Environmental Science, Policy, and Management
University of California, Berkeley, CA. 94720-3114
Phone 510-642-4934; email: bcollins@nature.berkeley.edu

Table of Contents

Introduction	1
Objective 1: Spatial patterns of large WFU fires	3
Objective 2: Comparing WFU fires to historical fires	15
Objective 3: Factors controlling self-limiting characteristics among WFU fires	22
Products	31
References	32

Introduction

Fire has played a vital role in the development of western U.S. flora and fauna. Prior to Euro-American settlement, fires occurred frequently in dry forest types throughout this region. Whether naturally-ignited or anthropogenic, fire served as a regular ecosystem process, driving landscape patterns of vegetation, and influencing wildlife distributions, nutrient cycling, and water availability. Evidence from tree ring analysis and other paleoecological studies suggests that drier, mid- to low elevation forest types, such as ponderosa pine (*Pinus ponderosa*) and Sierra Nevada mixed conifer, burned many times per century. These low to moderate intensity fires often burned for months at a time, and collectively affected very large areas.

In the twentieth century, policies of comprehensive fire exclusion effectively removed fire as an ecological process in many of these drier forested ecosystems. The effects of such policies are complex and tend to vary by forest type, but it is generally agreed that a century of fire exclusion has caused significant changes in the structure and function of many forest types. In the absence of relatively frequent fire, the amount of dead and downed organic material (stems, branches, leaves, and needles), or fuel, has increased. Additionally, more shade tolerant tree species, such as white fir (*Abies concolor*) in Sierra Nevada mixed conifer forests, have become established at historically unprecedented densities. These changes have homogenized the spatial patterns of trees and fuels in forests, creating a much more continuous forest structure across landscapes. This homogenized structure has a much higher probability of supporting larger and more intense fires. The increased risks of catastrophic wildland fire, and shifts in species composition and structure, are indicators that the systems today are significantly different than they were in pre-settlement times. These changes threaten many of the values that we obtain from our wildlands, including timber and aesthetics, along with water and air quality.

As these adverse effects associated with fire exclusion came to light in the latter half of the twentieth century, managers and scientists began to realize that fire should be returned to the landscape in some capacity in order to restore and maintain ecosystems. These pioneers were able to persuade the U.S. National Park Service and the U.S. Forest Service to shift their fire policies from total fire suppression, to a more comprehensive fire management strategy. This meant that both naturally-ignited and management-ignited fires could be used to achieve objectives aimed at restoring forest structure and re-establishing fire as an ecosystem process. However, there was substantial opposition and institutional dogma that proved difficult to overcome in attempting to implement this alternative fire management strategy. As such, there were only a handful of areas that actually used fire regularly as a management tool.

In the early1970s, the National Park Service introduced the Prescribed Natural Fire program (now referred to as Wildland Fire Use – WFU) in two Sierra Nevada wilderness areas, the Sugarloaf Creek basin in Illilouette Creek basin in Sequoia/Kings Canyon and Yosemite national parks, respectively. Under this program, naturally ignited fires have been allowed to burn relatively unimpeded across each respective landscape. This program continues to date in these areas, and in fact has grown substantially throughout the western U.S. As managers and policy makers increasingly recognize the role of fire in forested ecosystems, along with the social and economic benefits associated with burning, WFU is gaining traction as a fire management alternative.


View looking east from the top of Sugarloaf Mountain, Sequoia National Park (photo by C. Tetchman)

Little information exists, however, about the effects of WFU programs on a landscape level. It is hoped that the reintroduction of “natural” fires will create a patchwork of areas with varying fuel loads. This diverse landscape mosaic should have the effect of limiting the spread of fire as it burns from areas of high fuel loads to areas of low fuel loads. Thus, fires should become, to some degree self-regulating in their size and intensity. After over thirty years of WFU policy in areas such as the Sugarloaf Creek and Illilouette Creek basins significant opportunity exists to evaluate the effectiveness of these programs. Existing extensive Geographic Information System (GIS) databases, containing information on the location and extent of past fires, provided a unique vehicle for analysis.

The primary goal of this project was to determine, on a landscape scale, how effective the WFU programs in Sugarloaf Creek and Illilouette Creek basins have been at re-introducing fire and restoring more natural fire-vegetation dynamics. We set several objectives in attempting to achieve this goal: 1) explicitly characterize the pattern of burning in recent large WFU fires in each basin to assess not only the factors driving observed patterns, but the impact of fire in shaping vegetation across the each landscape; 2) reconstruct historical fire occurrence using fire-scars recorded in tree rings to compare WFU fires (from 1973-present) to historical fires; 3) use GIS-based maps WFU fires to investigate the extent that fires, when left to burn relatively freely, become self-limiting. In the following we will provide succinct descriptions of how we addressed each of these research objectives, along with explanations of our findings. More detailed information on the particular study designed to investigate each objective can be found in the peer-reviewed journal articles listed in the Products section of this report.

Objective 1:
Spatial patterns of large WFU fires in the Illilouette Creek and Sugarloaf Creek basins

The intent in studying this objective 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 (Figure 1-1). Each park is over 300000 ha and extends from the foothills (~500m 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 100cm. (Caprio and Graber 2000; van Wagtendonk et al. 2004).

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 1400 ha. Table 1-1 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 1-2.

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

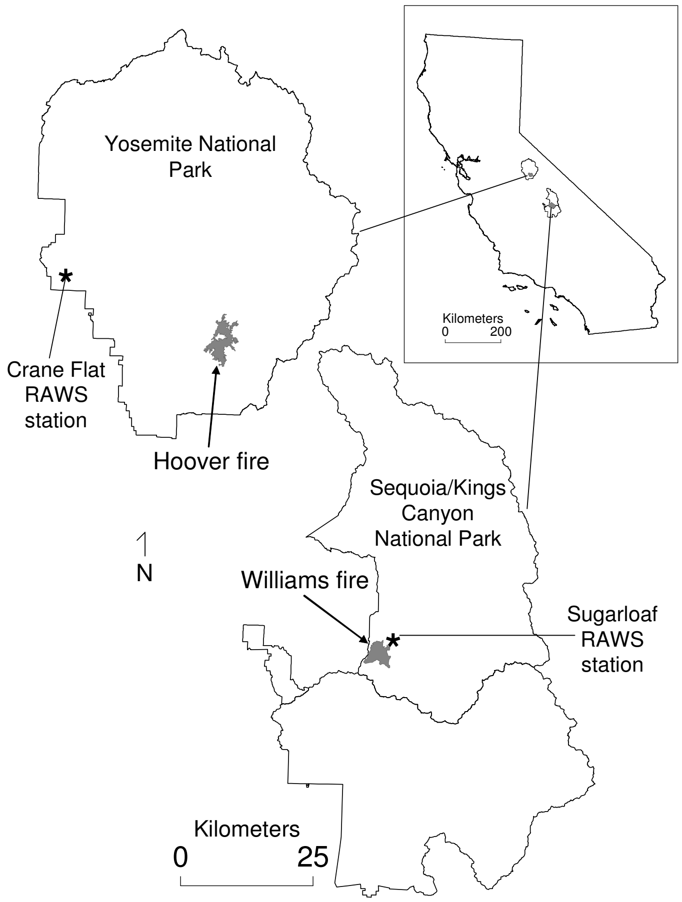


Table 1-1. Area burned in Hoover and Williams fires summarized by dominant vegetation type.

	Hoover Fire (2001) Yosemite National Park			Williams Fire (2003) Sequoia/Kings Canyon National Park		
	Number of cells (30 x 30m)	Hectares	%	Number of cells (30 x 30m)	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 1-2. 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

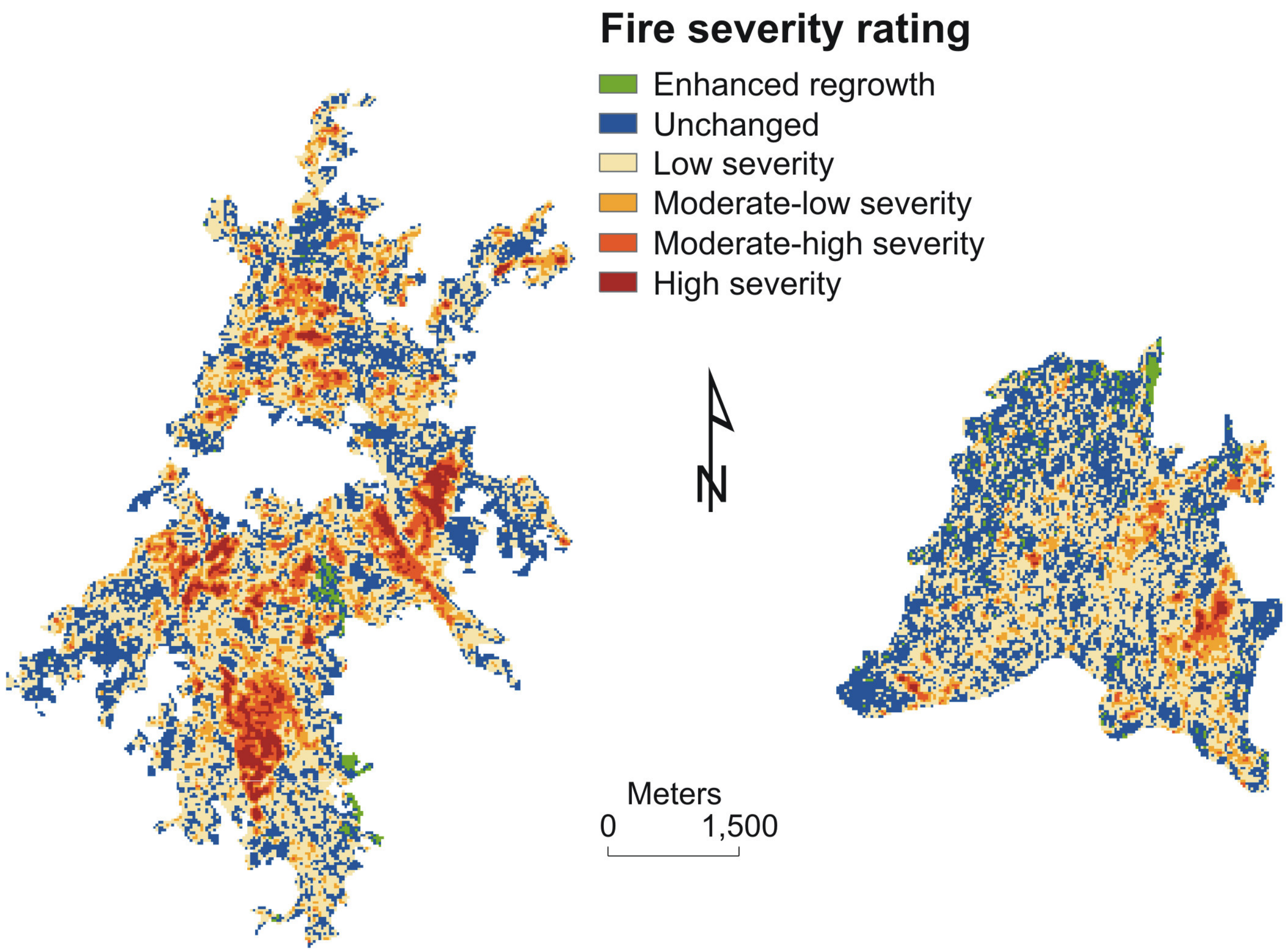
Half Dome (left), Mt. Starr King (right) and fire effects from 2004 WFU fire in the Illilouette Creek basin, Yosemite NP (photo by B. Collins)

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 30m spatial resolution, and import directly into a Geographic Information System (GIS). Figure 1-2 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 (30m) in both dNBR images we assigned values for vegetation type, weather, topography, and previous fire history. The vegetation type values were based 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 1-1. We use dominant vegetation type under the assumption that fuel amounts and fuel structure corresponded with vegetation type. 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 Figure 1-1). We averaged hourly values of temperature, wind speed gusts, and relative humidity to get daytime (10am-5pm) 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. 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. We assume that averaging

Figure 1-2. The differenced Normalized Burn Ratio images for the Hoover (A) and Williams (B) fires classified into fire severity ratings.



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 30m 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 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 30m 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 (1-4 30m cells) that account for over half the total number of patches in each severity class.

Statistical analyses

I 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 1000m for the Hoover fire and 750m 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 (Figure 1-3). The maps of fire severity (Figure 1-2) 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 (Figure 1-3) 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 (Figures 1-2 and 1-3).

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 (Figure 1-4). The lowest dNBR values corresponded 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 (Figure 1-5). 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 1-1).

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 (Figure 1-5). In the Williams fire, the highest fire severity occurred in Jeffrey pine and lodgepole pine stands when 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).

Figure 1-3. 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).

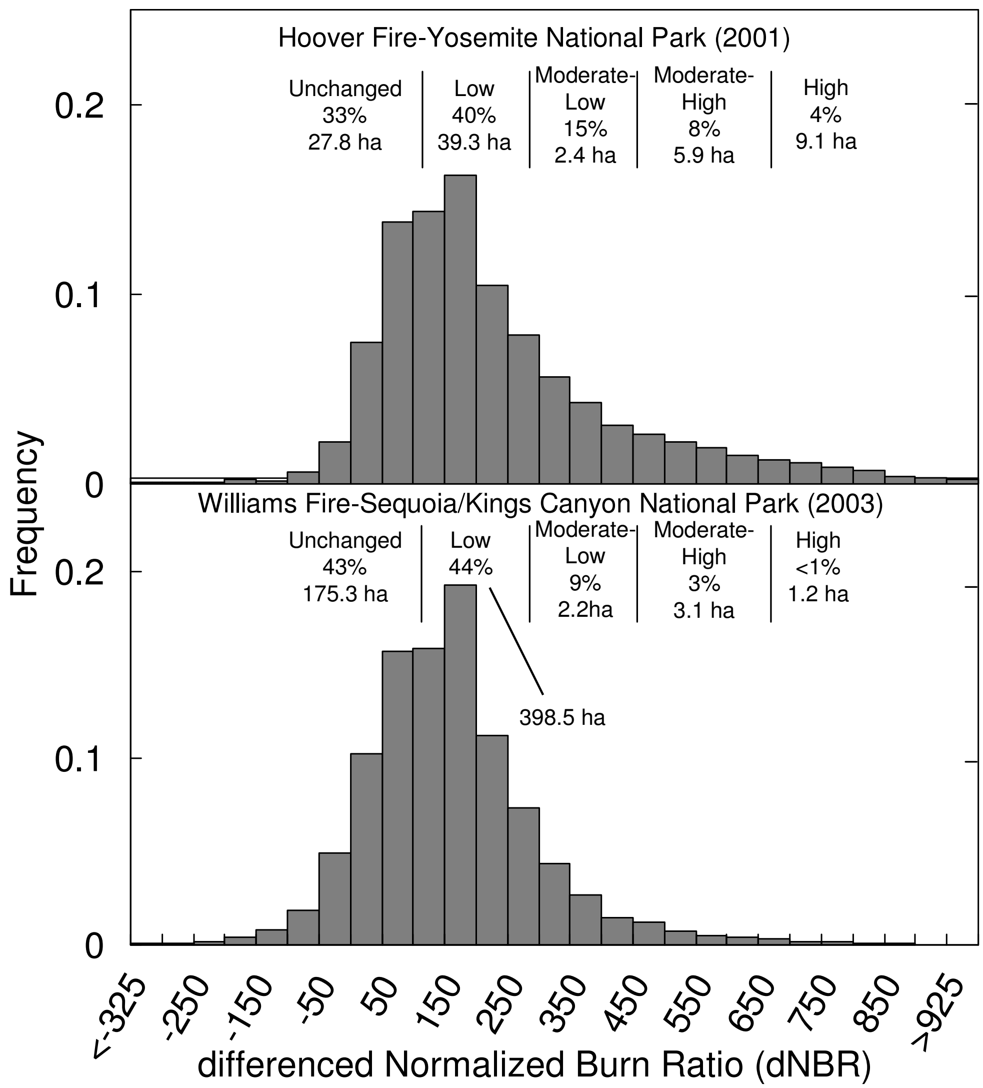


Figure 1-4. Regression tree explaining the spatial distribution of differenced Normalized Burn Ratio (dNBR) values throughout the Hoover fire. The number of pixels, along with the average dNBR, in the resulting group is reported at each node. The length of the line from each split indicates the relative proportion of total sum of squares explained by that split. The total R^2 for the tree is 14%.

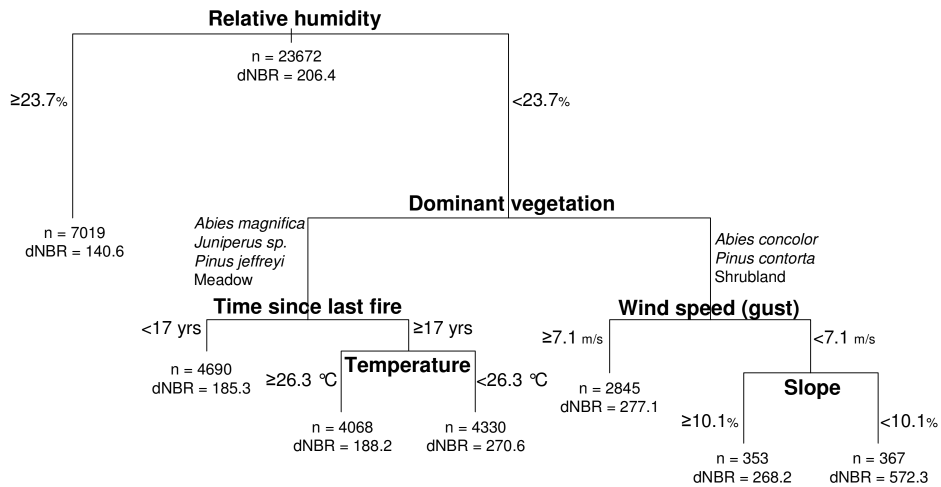
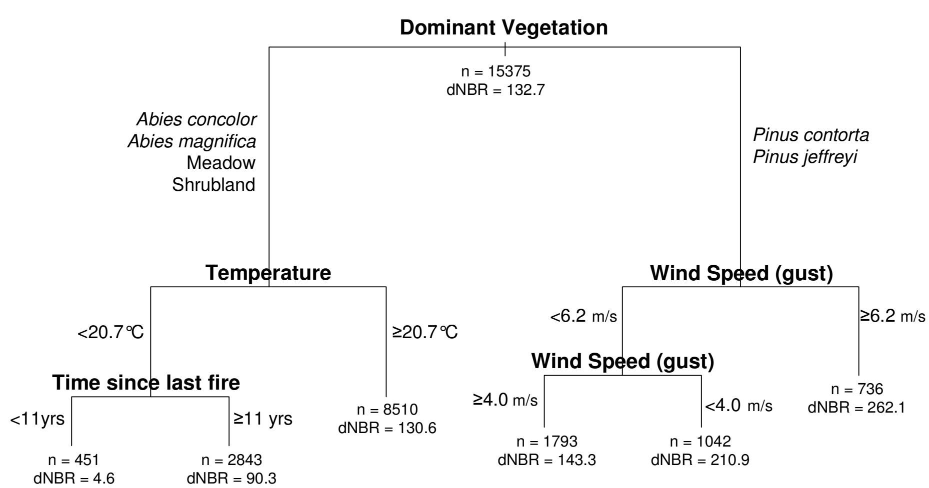


Figure 1-5. Regression tree explaining the spatial distribution of differenced Normalized Burn Ratio (dNBR) values throughout the Williams fire. The tree is drawn and labeled as in Figure 5. The total R^2 for the tree is 12%.most important explanatory variable in both fires.



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 (Figure 1-2). 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 (Figure 1-3). 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 (Figure 1-2). 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 1-1). 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 (Figure 1-2) and resulting patches of high and low severity between the two fires (Figure 1-3). 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.

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 (Figures 1-5 and 1-6). Summary

An example of variation in fire severity within a Jeffrey pine stand in a 2004 WFU fire, Illilouette Creek basin, Yosemite NP (photo by B. Colllins)

statistics for weather indicate generally higher temperatures and wind speeds during the Hoover fire, but lower relative humidity during the Williams fire (Table 1-2). 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.

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.

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 (Hessburg et al. 2005; Miller and Urban 2000a). 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.

Objective 2:
Comparing WFU fires to historical fires in the Illilouette Creek and Sugarloaf Creek basins

Here we present reconstructions of historical fire occurrence using tree ring proxies, along with chronologies of tree recruitment to make inferences on the effects of WFU programs on forest structure. Our objectives are to compare the frequency and extent of WFU fires to that of historical (pre-fire suppression) fires in Illilouette Creek and Sugarloaf Creek basins. Additionally, we aim to investigate the impact of the fire exclusion period on tree recruitment, relative to the historical time period and the WFU period in both basins. The term "tree recruitment" refers to trees that established and have persisted to the present (sensu Brown and Wu 2005).

Methods

Within each basin we designated an approximately 500 ha study area to investigate fire history and stand age structure (subsequently referred to as just Illilouette and Sugarloaf for brevity) (Figure 2-1). The locations of these study areas were chosen to optimally capture the range of area burned at different frequencies by WFU fires. We stratified the study areas by burn frequency (0 – 4 burns since beginning of natural fire programs - 1972) then established a 200 m grid for stand/age structure sample plot locations. In Sugarloaf we had to use a 100 m grid for the 0 burn frequency stratum because very little unburned area existed. We intended to sample five plots in each burn frequency stratum, but more importantly we wanted to get ages from approximately 250 trees in each study area to study regeneration structure. A total of sixty-three 0.05 ha circular plots were sampled between study areas, 24 in Illilouette and 39 in Sugarloaf. In each plot we extracted increment cores from every live tree ≥10 cm diameter at breast height (dbh). Trees were cored, sometimes repeatedly, at approximately 20 – 30 cm above the ground level until we reached pith or no more than a field-estimated 10 years from pith.

We opportunistically collected cross-sectional slabs from 73 fire-scarred trees, snags, and downed logs. We cut and removed slabs from trees within approximately 70 m of plot center exhibiting visual evidence of multiple fire scars. We also collected slabs from any tree we noticed with multiple fire scars while walking between plots. Sampling only multiple-scarred trees results in an efficient method for detecting the maximum number of fire years with the least ecological damage and field/laboratory work (Brown and Wu 2005). Furthermore, recent research has shown that opportunistic or "targeted" sampling of fire-scarred trees to reconstruct historical fire occurrence yields results very similar to either random or systematic sampling (Van Horne and Fule 2006). Tree cores and fire-scarred slabs were sanded to a high polish, then crossdated against a master chronology using standard dendrochronological techniques to assign calendar years to pith dates and fire scars (specific methodology was as explained in Brown and Wu 2005).

We limited the starting year of the analysis to 1700 because there was insufficient sample depth in the fire scar record prior to that period (Figure 2-2). we mapped the spatial extent of fires recorded within our study area by constructing a polygon around trees (3 minimum) that recorded a particular fire (Bekker and Taylor 2001). While a complete census of all fire-scarred trees would be optimal for the most accurate reconstruction of fire extent, doing so would be

Figure 2-1. Digital elevation models, stand/age structure plot (gray circles), and fire-scarred tree (black circles) locations in Illilouette Creek basin (upper) and Sugarloaf Creek Basin (lower), California, USA.

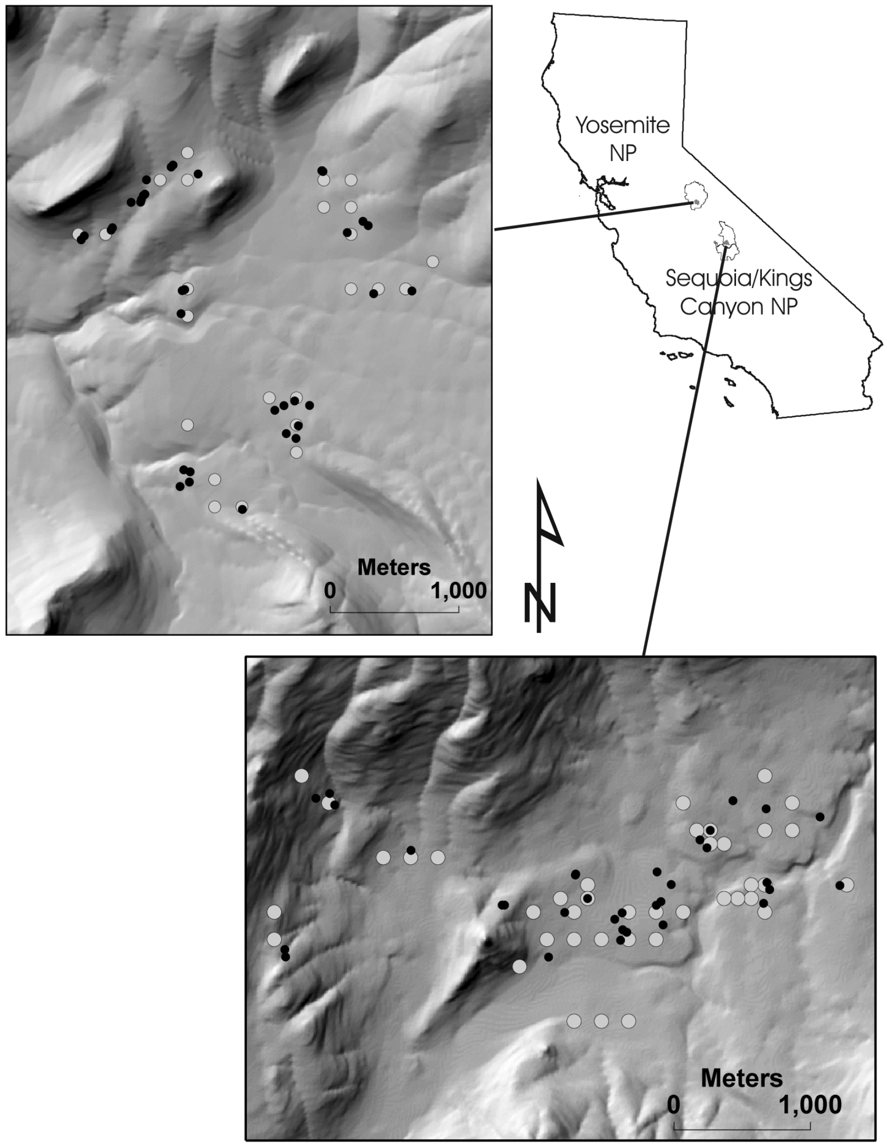
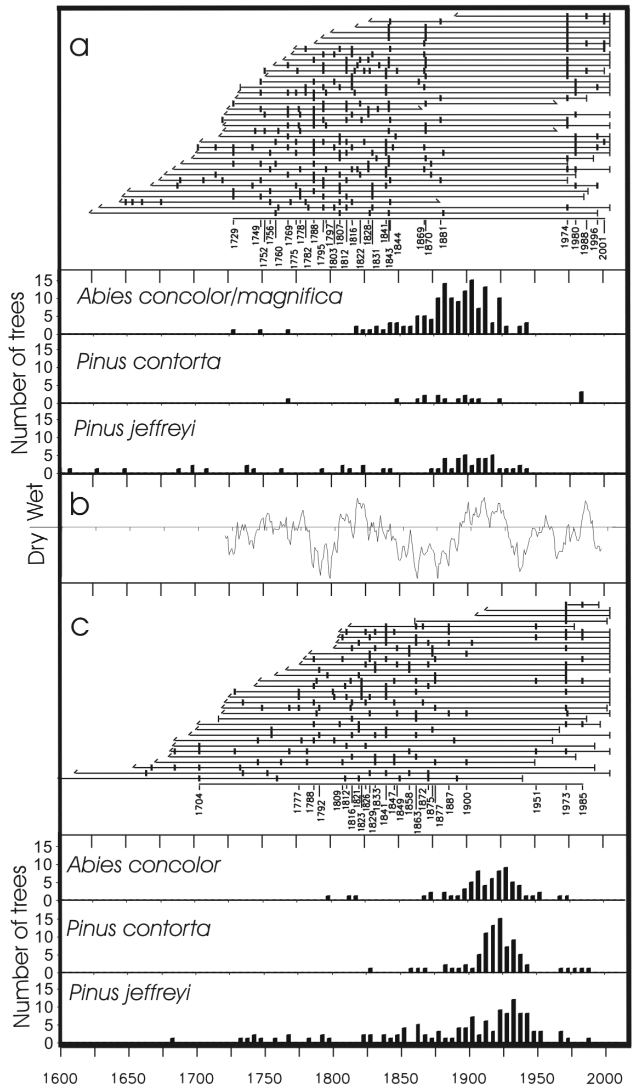


Figure 2-2. Fire year and tree recruitment chronologies for Illilouette Creek (a) and Sugarloaf Creek (c) basins. Horizontal lines represent the time span of individual fire-scarred trees, with dark vertical tick marks marking fire scars. Listed below are fire years in which ≥ 3 trees were scarred. Solid vertical bars below each fire chronology are tree recruitment dates by 5 year periods. (b) 20 year moving average of reconstructed Palmer Drought Severity Index for grid point 14 (Cook et al. 1999).



infeasible due to the effort required and destructive nature of sampling fire scars, especially in wilderness areas. Based on the relatively complete coverage of each study area we feel that we were able to sufficiently reconstruct fire extent (Figure 2-1). To investigate this assertion we employed a species accumulation approach towards analyzing fire years. We constructed accumulation curves by plotting the average number of fire years detected as a function of the number of fire-scarred trees sampled. The averages were calculated from 100 permutations, in which fire-scarred trees were added in random order up to the total number of trees for each study area (n = 33 and 37 for Sugarloaf and Illilouette, respectively). Michaelis-Menton saturating functions were fit to each curve to estimate the asymptote, or theoretical maximum number of fire years for each study area (Battles et al. 2001). The accumulation curves indicate that in both study areas the total number of fire years detected is within 20% of the theoretical maximum for Illilouette and 26% for Sugarloaf. We feel this discrepancy is reasonable given that nearly half of all fire years detected in both areas appear to be localized 'spot' fires that scarred two or less trees and therefore, we have probably detected all wide-spread fires.

Examples of 'catfaces' caused by repeated fire at the base of these Jeffrey pines in Sugarloaf Creek basin (photo by C. Tetchman)

Fire scarred cross-section that was removed with a chainsaw, sanded to a high polish, and cross-dated to determine historical fire dates (ph. by B. Collins)

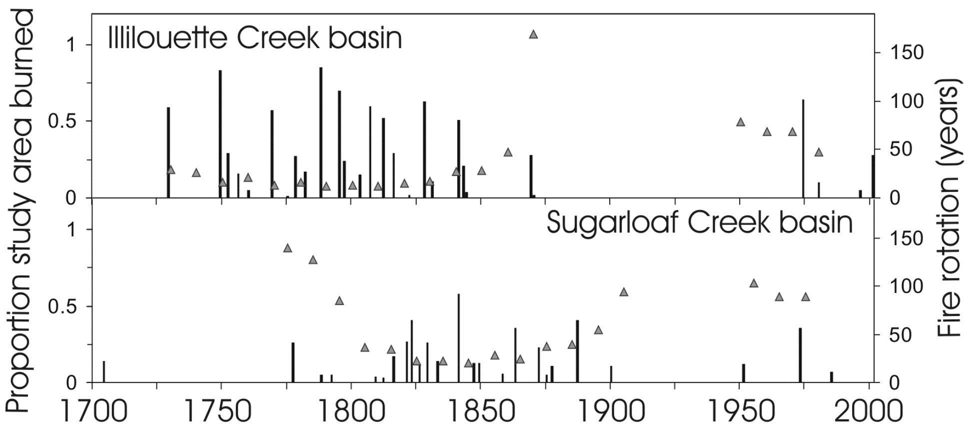
Results

We were able to establish pith dates for 489 trees and successfully crossdate 420 fire scars from 70 trees between Illilouette and Sugarloaf (Figure 2-2). We could not determine pith dates on 22 of 239 trees in Illilouette and 28 of 300 trees in Sugarloaf. In Illilouette, white fir was the most abundant tree species (60%), followed by Jeffrey pine (26%), lodgepole pine (8%), and red fir (6%). The mix of tree species was more even in Sugarloaf, with Jeffrey pine being most abundant (44%), followed by lodgepole pine (31%), and white fir (25%). In both areas the oldest trees were Jeffrey pine, which exceeded 300 years old, while white/red fir and lodgepole pine seldom exceeded 200 years old. White/red fir recruitment, and to a lesser extent lodgepole pine and Jeffrey pine recruitment, noticeably peaked between 1875 and 1920 in Illilouette. In Sugarloaf, a similar peak in recruitment is evident for all three species; however, this peak occurred later, between 1895 and 1940 (Figure 2-2).

The earliest recorded fires were 1650 for Illilouette and 1665 for Sugarloaf. Fires occurred fairly frequently throughout the reconstructed period of record (Figure 2-2). Using a minimum criterion of at least 3 trees recording a given fire year, the mean fire return interval (MFI) between 1700 and 1900 was 6.3 years for Illilouette and 9.3 years for Sugarloaf. Using the same minimum criterion and time period the fire rotation (defined as the length of time necessary to burn a cumulative area equivalent to the size of the study area) was 24.7 years for Illilouette and 49.2 years for Sugarloaf. This difference is due to the generally smaller fires in Sugarloaf (Figure 2-3). Fires clearly stop occurring after 1881 in Illilouette and 1904 in Sugarloaf, with the last fires of any substantial spatial extent occurring in 1869 and 1900 in Illilouette and Sugarloaf, respectively (Figures 2-2 and 2-3). With the exception of the 1951 escaped wildfire in Sugarloaf, fires do not occur again until the start of WFU policies. For the WFU period (1972 – present) we detected five fires, with an MFI of 6.8 years in Illilouette, while only two fires in Sugarloaf, with 12 year interval between the fires. Fire rotations during this period were 32.9 and 79.7 years for

Illilouette and Sugarloaf, respectively.

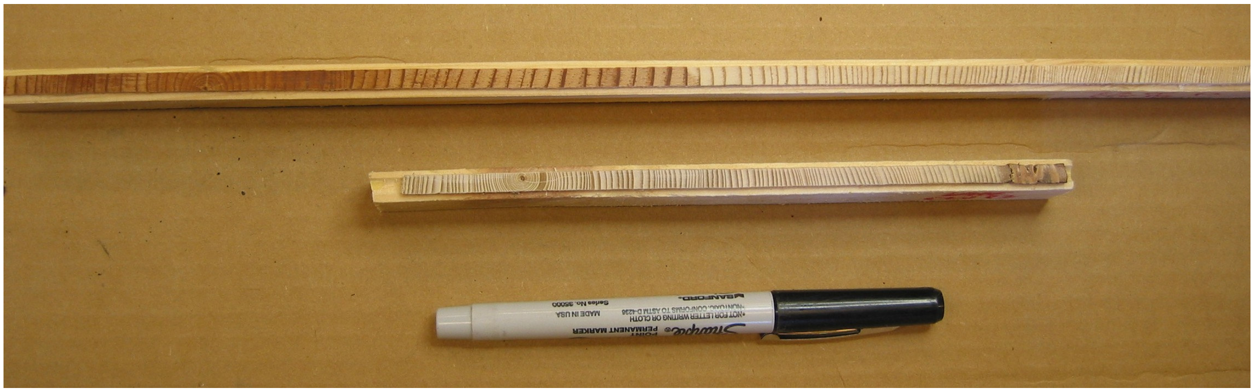
Figure 2-3. Reconstructed fire extent (vertical bars) and fire rotation (gray triangles), within each study area, for years in which ≥ 3 trees were scarred. Fire rotation is defined as the length of time (years) necessary to burn a cumulative area equivalent to the size of the study area, and is calculated at 10 year intervals for overlapping 50 year periods.



Discussion

As with many of the dry forests throughout the western U.S., fire historically played a major role in shaping and maintaining the forests in Illilouette and Sugarloaf (Figure 2-2). The frequency at which fires burned, along with the persistence of many of the older trees (mostly Jeffrey pine) suggests that fires in the 200 years prior to fire exclusion burned typically under moderate- to low-intensities. Based on the data presented here there is little evidence to suggest WFU fires burned differently. The frequency and extent of fires during the current WFU period approaches that of historical levels, especially in Illilouette (Figure 2-3). In fact, analysis of recent large WFU fires in both Illilouette Creek and Sugarloaf Creek basins demonstrates that spatial patterns of fire-induced mortality are similar to our understanding of such patterns for historical fires (Collins et al. 2007).

Similarities between historical fires and recent WFU fires are surprising given the obvious changes in tree recruitment that coincided with the fire exclusion period (Figure 2-2). In both Illilouette and Sugarloaf, unprecedented peaks in tree recruitment began shortly after the last extensive fire in each area (1869 in Illilouette, 1887 in Sugarloaf). The cessation of fires, along with a shift towards wetter climatic conditions ca. 1900, most likely explains the observed pulses in tree recruitment (Figure 2-2) (Brown and Wu 2005). Historically, frequent fires moderated tree recruitment by killing small trees that did not grow to a point at which trees could resist fire (i.e. trees crowns were not high enough off the ground or bark was not thick enough to insulate tissues from thermal damage) (North et al. 2005; Stephens and Fry 2005). The long fire free interval resulting from fire exclusion allowed for increased establishment and growth beyond this vulnerable stage for all three dominant tree species in Sugarloaf, and mostly white fir in Illilouette. While we do not know the extent to which trees were killed by WFU fires, especially early larger WFU fires, we do know that many survived these fires and persist through the present.



Extracted, mounted, and sanded tree cores from an old Jeffrey pine (top), and a relatively young white fir (bottom) (photo by B. Collins)

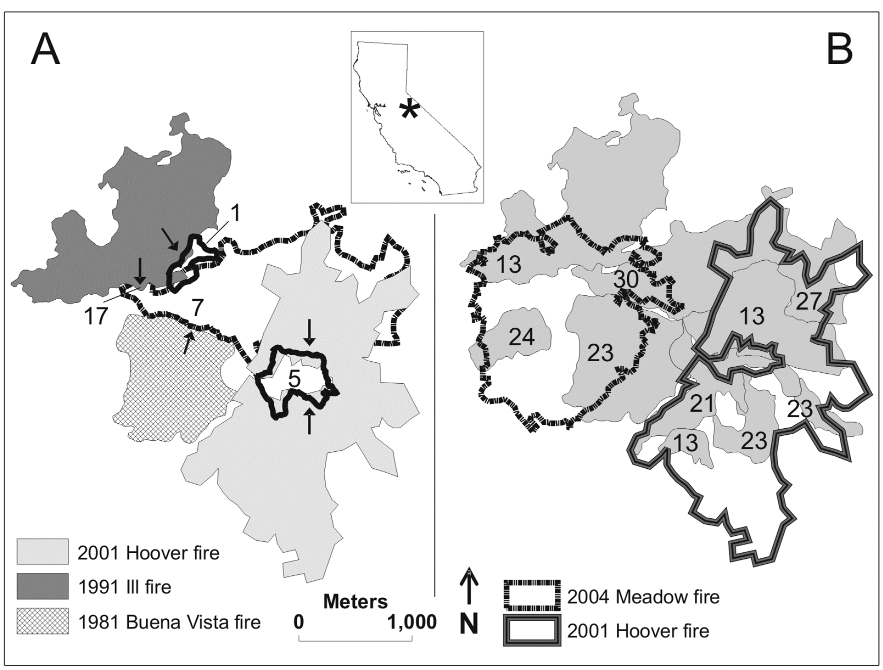
Given that so many of the trees which became established during fire exclusion survived multiple WFU fires, it seems they will continue to persist for some time (Miller and Urban 2000b), barring the occurrence of much more severe fires in the future. While, it is possible that competitive interactions among these trees have, and continue to influence forest structure, we submit that fire is the dominant process driving forest structure in these forests. As such, perhaps restoring historical forest structure by fire alone is infeasible for the near future. However, what may be more important than restoring structure is restoring the process, fire (Stephenson 1999). By allowing the process of fire to resume its natural role in limiting density and reducing surface fuels, competition for growing space is reduced, as well as potential fire severity in subsequent fires (Fule and Laughlin 2007). As a result, we contend that the forests in Illilouette and Sugarloaf are becoming more resistant to ecosystem perturbations (e.g. insects and disease, drought, etc.). This resistance could be important in allowing these forests to cope with projected changes in climate.

Ultimately, active restoration resembling historical forest structure across landscapes may be undesirable considering the effort required to manipulate tree density and fuels. Furthermore, the climate of the historical period is a poor surrogate for future climates. However, there is clearly a need to incorporate fire into the management of drier, historically more open forest types throughout the western U.S. The results from this study show two clear examples in such forest types where the process of fire has successfully been returned over each respective landscape. Although it is not ubiquitously applicable, WFU can potentially be a cost-effective and ecologically sound tool for ‘treating’ large areas of forested land. Decisions to continue to suppress fires are politically safe, but ecologically detrimental. Each time the decision to suppress is made the risk of a fire escaping and causing damage (social and economic) is essentially deferred to the future. Allowing more natural fires to burn under certain conditions will most likely mitigate these risks. It is imperative that the public recognizes this and becomes more tolerant of some direct consequences (i.e. smoke production, limited access, etc.).

Objective 3:
Factors controlling self-limiting characteristics among WFU fires

In this study, we investigate the extent to which wildfires can be 'self-limiting', and if so, what environmental conditions govern the limiting interactions among fires. If wildfires are limited in extent by previous fires, then increased burning (both with prescribed fire and allowing natural wildfires) can be a viable solution in mitigating potential increases in uncharacteristically severe fires. If not, intentionally burning, or allowing wildlands to burn naturally may be an ineffective strategy of alleviating wildfire hazard. The Illilouette Creek basin provides a relatively unique opportunity to investigate the interactions among wildfires that burned relatively unimpeded across the landscape for over three decades. The goal of this study is to investigate the interactions between successive wildfires, and assess to what extent the environments in which they burn influence these interactions. We define an interaction where a fire burned up to, but not over a previous burn, as extent-limited (Figure 3-1A), and where a fire burned over a previous burn perimeter, as a reburn (Figure 3-1B). Additionally, we investigate potential trends in post-fire effects on the dominant vegetation, or burn severity, for these naturally-occurring fires over the last three decades. we analyzed mapped fire perimeters (Morgan et al. 2001), along with recently developed satellite-based estimates of burn severity (Miller and Thode 2007; Thode 2005), to characterize the attributes of, and identify interactions among 19 fires that burned between 1974 and 2004 in the Illilouette Creek basin.

Figure 3-1. Examples where wildland fires were limited in extent by previous fires (A) and where fires burned over previous fires (B) in the Illilouette Creek basin, California, USA. The numbers within each fire outline indicate the time (years) since each previous fire. The arrows (A) show where each listed fire was limited in extent by a previous fire.



Methods

Study area

Illilouette Creek basin is in the central Sierra Nevada, CA, USA (Figure 3-1). The basin is over 15,000 ha with elevations ranging from 1400 m 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°C to -5°C, while average July maximum temperatures range from 24°C to 32°C. Precipitation varies with elevation and is predominantly snow, with annual averages near 100cm. The forests in Illilouette Creek basin are dominated by Jeffrey pine (*Pinus jeffreyi*), lodgepole pine (*Pinus contorta*), white fir (*Abies concolor*), red fir (*Abies magnifica*), and are interspersed with meadows and shrublands.

Spatial and weather data

We analyzed mapped fire perimeters (Morgan et al. 2001), along with recently developed satellite-based estimates of post-fire effects (Miller and Thode 2007; Thode 2005), to characterize the attributes of, and identify interactions among 19 fires that burned between 1974 and 2004. Some larger fires had extent-limited and/or reburn interactions with multiple fires, giving us a total of 30 interactions (14 extent-limited, 16 reburn). To characterize the environmental conditions that influence fire behavior, and likely could affect interactions among fires, we compiled spatial datasets for each fire that capture the physical environment (topography, dominant vegetation type) in which they burned, as well as the weather conditions during burning.

We obtained digital fire perimeters (fire atlases) from Yosemite National Park personnel for each WFU fire that occurred between 1974 and 2006. The fire atlases are a best approximation of actual burn perimeters, but do not provide information of spatial patterns within burn areas (Morgan et al. 2001). We used satellite based estimates of burn severity to assess spatial patterns of burning. Based on the availability of satellite imagery, burn severity data for fires prior to 1984 was derived using a relative version of the difference between pre-fire and post-fire Normalized Difference Vegetation Index (RdNDVI) (Thode 2005). This index indicates changes in reflectance resulting from consumption of vegetation, burned vegetation, and charred soil. For fires that occurred in 1984 or later, we used a relative version differenced Normalized Burn Ratio (RdNBR), which improves the estimation of burn severity across multiple fires and vegetation types (Key and Benson 2005; Miller and Thode 2007; Thode 2005).

We used fire atlases and burn severity images to identify extent-limited (Figure 3-1A) and reburn (Figure 3-1B) interactions. The rule used to define extent-limited interactions was where two fires shared a common border that was at least 10% of the more recent fire's perimeter, and less than 200 m overlap in fire perimeters occurred. When interpreting common fire borders we visually inspected high-resolution air photos to identify landscape features that would act as natural firebreaks (rock outcrops, wet meadows, etc.). Extent-limited interactions that could have been influenced by natural firebreaks were not included in the analysis. Reburn interactions were defined as instances where a fire burned more than 200 m into the area burned by a previous fire.

Using ArcGIS software (ESRI 2004) we assembled datasets for dominant vegetation, slope gradient, and weather for each WFU fire. Rather than averaging across each fire, we sub-sampled the burn severity, slope gradient, and vegetation for the area immediately surrounding the interactions between fires. For each extent-limited interaction we created a 200 m buffer around the common border and extracted data for each of the spatial variables. For reburn interactions we only extracted data that was within the area burned over. In each case, we averaged the spatial variables over the sub-sampled area to get one value for each of the following variables for each fire interaction: burn severity in the previous fire, time since previous fire, dominant vegetation type, slope gradient, and an integrated index of potential fire behavior, the burning index at the time of the second fire. The burning index is strongly influenced by weather, but also incorporates topographic and fuel influences on fire spread and heat output. Since the fuel component of the burning index calculation is based on relatively coarse-scale estimates of fuel structure and abundance, or fuel model, we only use one fuel model for the entire Illilouette Creek basin, and therefore we use the burning index as a proxy for fire weather (Bradshaw et al. 1983).

Daily burning index values were derived from temperature, wind speed, and relative humidity observations taken at Crane Flat weather station (available from http://famweb.nwcg.gov/). Crane Flat was the nearest weather station that had records back to 1974. we choose *a priori* to use 95th percentile burning index values rather than averaging daily burning index values for the entire duration of each fire, under the assumption that peak weather can often grossly affect fire behavior (Crosby and Chandler 2004). Averaging daily values over the duration of each fire (which was often 2 months or more) would reduce the influence of these peak weather periods.

An example of an un-manipulated fire edge in a 2004 WFU fire in the Illilouette basin, Yosemite NP (photo by B. Collins)

Statistical analyses

I modeled extent-limited and reburn interactions between fires using two techniques: (1) categorical tree analysis, which we used to identify potential threshold values for variables explaining interactions among fires; (2) logistic regression, which we used to more directly assess the strength of both individual predictor variables and the model as a whole (Breiman et al. 1984; Hosmer and Lemesow 2000). We treated each fire interaction as an independent observation. We feel this assumption is reasonable, even given that a single large fire could have multiple interactions with smaller previous fires, because each interaction involves a distinct combination of the predictor variables (burn severity in the previous fire, time since previous fire, dominant vegetation type, slope gradient, burning index). We also used the post-fire effects on the dominant vegetation, or burn severity data to investigate any potential trends in wildfire effects over the time span of these fires.

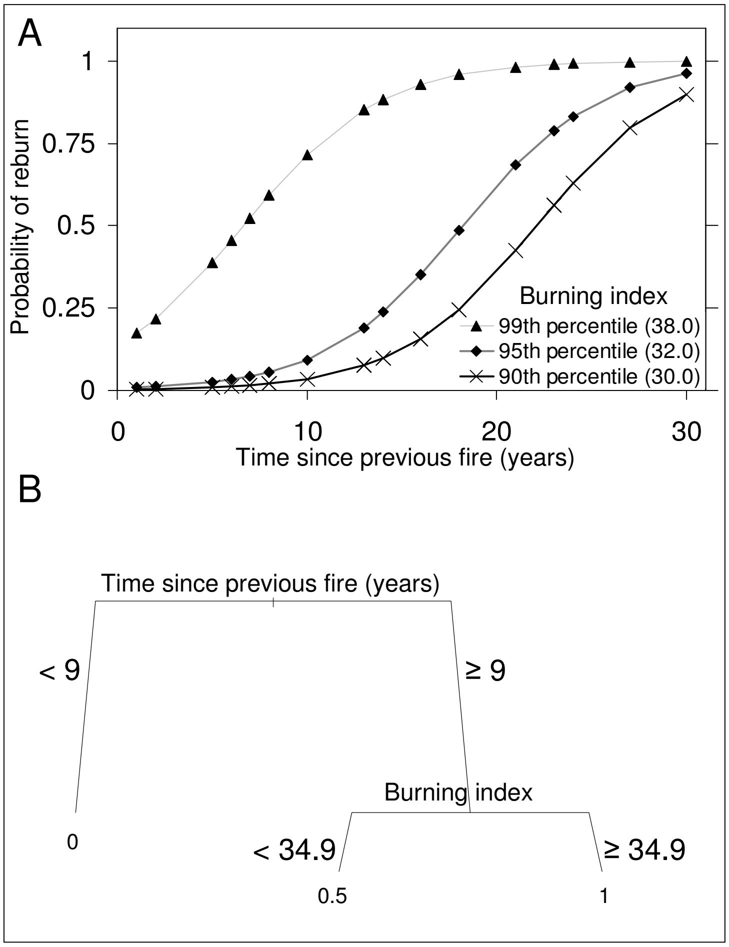
The categorical tree is constructed by repeatedly splitting the data into increasingly homogenous groups based on the response variable, extent-limited (0) or reburn (1) interaction. Each split is based on a simple rule for a given predictor variable ($\geq$ or $<$), which minimizes the sum of squares within the resulting groups. The number of splits was determined using the one-standard error rule on the cross-validated relative error (Breiman et al. 1984; De'ath 2002). The rule for each split identifies the value or level of a given predictor variable at which the response, which was probability of reburn, changes substantially. For logistic regression, we used a stepwise model selection method ($\alpha = 0.1$) with the same set of predictor variables mentioned previously. None of the variables exhibited any collinearity with other variables. we used a goodness of fit test to evaluate the adequacy of the logistic regression model ($P = 0.99$) (Hosmer and Lemesow 2000).

Results and Discussion

Both statistical techniques adequately modeled the interactions between fires (categorical tree $R^2 = 0.73$ (apparent), 0.39 (relative); logistic regression area under ROC curve = 0.93). Furthermore, both techniques identified the same two predictor variables as being important in explaining extent-limited versus reburn interactions between fires: time since previous fire (logistic $P = 0.07$) and the burning index (logistic $P = 0.05$) (Figure 3-2). The probability of an interaction between two fires resulting in a reburn increases as time since previous fire increases, and as fire weather becomes more extreme, i.e. higher wind speed, lower relative humidity, and higher air temperature (Figure 3-2A). Nine years since the previous fire appears to be a threshold at which previous fires have less of a limiting effect on the extent of subsequent fires (Figure 3-2B). Under 9 years, the categorical tree analysis indicates a zero probability of reburn, and a 100% probability of an extent-limited interaction. The logistic regression analysis similarly demonstrates very low probabilities of reburn for high and moderate fire weather conditions when time since previous fire is below 9 years. However, under extreme fire weather conditions, the effect of time since previous fire on the probability of reburn is substantially reduced (Figure 3-2A). Neither model identified dominant vegetation type, slope gradient, individual weather parameters (temperature, wind, humidity), or the burn severity in the previous fire as influencing the interactions between fires.

Forests in the Illilouette Creek basin historically burned relatively frequently, at moderate – to low-intensities. Recent studies have shown that the WFU fires in the last 33 years did not

Figure 3-2. Logistic regression probabilities under different weather scenarios (extreme – 99th, high – 95th, and moderate – 90th percentile burning index values) (A), and categorical tree break points (B) explaining the influence of both previous fires and weather (burning index) on extent-limited (0) and reburn (1) interactions among wildland fires. The length of the line from each split in the categorical tree indicates the relative proportion of total sum of squares explained by that split. The number below each terminal node in the categorical tree is the probability of scarring from WFU fires for that group.



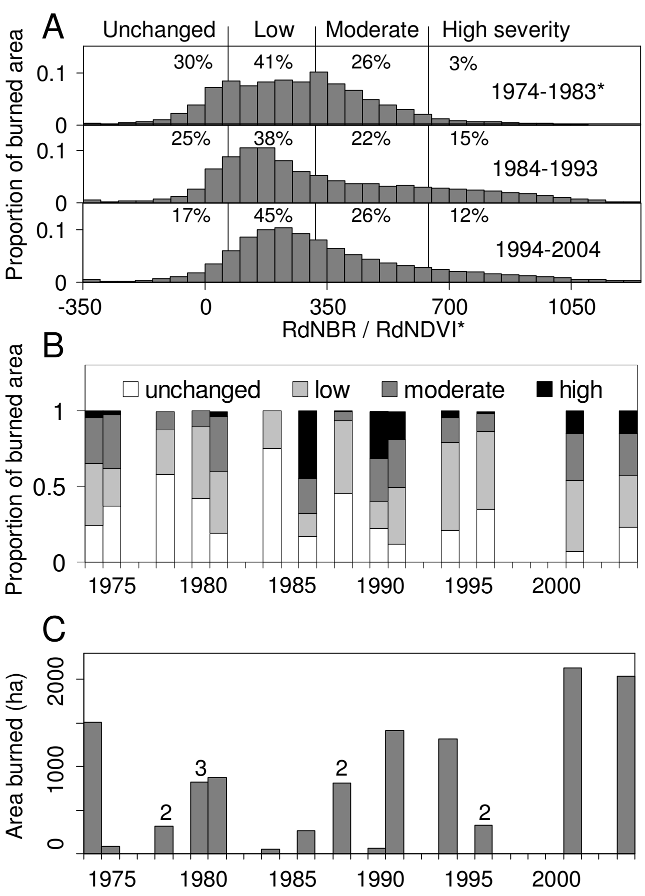
differ in extent, frequency, or effects on vegetation from reconstructions and interpretations of historical fires (Collins et al. 2007; Collins and Stephens 2007). Fire regimes of this type are historically characterized as being limited by surface fuel amount and continuity (Schoennagel et al. 2004). These findings indicating the importance of time since previous fire in limiting the extent of subsequent fires support the idea of a fuel limited system. The 9-year threshold for time since previous fire can be interpreted as the time necessary for sufficient fuel accumulation, after which there is enough fuel to carry fire into previously burned areas. It is interesting that the burn severity in the previous fires, which could theoretically be linked to the amount of fuel consumed, was not important in explaining the interactions between fires.

The importance of weather in both statistical models demonstrates that fuel accumulation is not the only mechanism driving the interactions among WFU fires. Even at relatively short intervals between successive fires (10 years), the probability of reburn is very high (0.72) under extreme fire weather conditions (Figure 3-2A). The lower relative humidity, higher winds speeds, and elevated air temperatures associated with these conditions creates higher quantities of available fuel by desiccating fuels that would not readily burn under less extreme conditions. In addition, more extreme weather leads to increased fire intensity, which augments fuel desiccation by preheating fuels adjacent to the flaming fire front. The findings from this study suggest that more extreme fire weather creates conditions that may overwhelm the mechanism of fuel accumulation. It is important to note that the conditions modeled under the extreme fire weather scenario are the 99th percentile fire weather; this means that only one percent of the days throughout the entire 30-year study period during which these WFU fires burned had weather that was at or exceeded these extreme fire weather conditions. Not only are 99th percentile weather conditions rare, it is even more unlikely that these conditions would coincide with an ignition or with a fire already burning. Therefore, we submit that for a large proportion of potential fire interactions, fires will be limited in extent by previous fires that burned relatively recently (<9 years).

Smoke production from the 2001 Hoover fire (Illilouette basin) burning under high to extreme fire weather (photo by E. Duncan)

This 'self-limiting' effect could potentially break down if extreme weather conditions occur more frequently in the future. We did investigate for evidence of trends in burn severity throughout the period of these WFU fires that could indicate climate-influenced changes in wildfire activity (Figure 3-3A). Using established thresholds for burn severity classes (Miller and Thode 2007) we tested for differences in the proportions of area within each severity class among decades. When all three decades were compared together a difference was apparent (Chi-square $P = 0.05$). This difference was most likely driven by the much lower proportion of area burned at high severity, and to a lesser extent the higher proportion of area unchanged, in the first decade of WFU fires (1974-1983). When examined along with the yearly proportion of area burned in each burn severity class (Figure 3-3B) it is clear that fires since the mid-1980s have tended to burn more severely than those that occurred in the first decade of WFU. The change in burn severity in the mid-1980s is consistent with the timing of substantial increases in large wildfire frequency throughout the western U.S. (Westerling et al. 2006). The fact that four of the five largest fires recorded in the WFU period occurred in 1991 or later further supports increased large wildfire frequency in the last decades (Figure 3-3C).

Figure 3-3. (A) Frequency distributions of burn severity pixels (relative differenced normalized burn ratio – RdNBR) for each decade since the beginning of the natural fire program in Illilouette Creek basin, California, USA. 2004 was included in the previous decade because it was the only fire in the recent decade. (B) Yearly proportion of area burned in each burn severity class. (C) Yearly area burned within the basin. The number on top of the bars indicates the number of fires that contributed to the burned area for a given year. No number means the area burned was from a single fire. *Prior to 1984 the calculations for burn severity (using the relative difference in normalized differenced vegetation index – RdNDVI) were slightly different due to limitations in the satellite imagery (Thode 2005). RdNDVI values were re-scaled so that they could be plotted on the same axis as the RdNBR values.



One factor that could confound our assertion of increasing burn severity is that the severity data from 1974-1983 were derived from an earlier satellite-based sensor (Landsat MSS) that had fewer bands than the post 1983 data (derived from Landsat TM). Based on differences in the number of bands in each sensor, different algorithms were used to compute burn severity estimates. However, comparisons of the two algorithms for the same fires using the Landsat TM imagery demonstrate a relatively consistent classification of burn severity (Thode 2005). Therefore, we submit that the observed changes in proportion of area burned under high severity reflect actual differences in post-fire effects, rather than methodological inconsistencies.

While the increase in area burned under high severity around the mid-1980s is apparent, there is no evidence for an increasing trend since then (Figure 3-3B). There was no statistical difference in the proportions of area within each severity class between the 1984-1993 and 1994-2004 periods (chi-square $P = 0.43$). Despite regional increases in wildfire activity throughout the western U.S. (Westerling et al. 2006) the effects of the WFU fires in the Illilouette Creek basin have remained relatively constant over the last 22 years. This consistency in burn severity may be a reflection of a 'self-limiting' process that not only limits the extent of fires, but also the effects of fire on vegetation.

Low- to moderate-severity effects four years following the 2001 Hoover fire, Illilouette basin, Yosemite NP (photo by B. Collins)

Our analysis of the interactions among relatively unmanaged wildfires offers new insight into the controls operating on fire at a landscape scale. When fires were allowed to burn over the last three decades, they became ‘self-limiting’, even under high fire weather conditions. The implication of these findings is that more Wildland Fire Use in similar dry forest types throughout the western U.S. may be a solution to mitigating anticipated increases in wildfire extent and effects (Kitzberger et al. 2007; Westerling et al. 2006). These results indicate that if increased burning is done at landscape scales, fires will not only limit future fire activity, they have a high likelihood of achieving restoration-based objectives (Collins et al. 2007; Fule and Laughlin 2007). The fact that our analysis did not identify burn severity in prior fires as contributing to the explanation of interactions with subsequent fires demonstrates that the burning characteristics of a fire are not nearly as important as the time since fire.

We do recognize within the dry forest types throughout the western U.S., increased intentional burning is not entirely applicable. As human communities expand into the wildlands, and as smoke production from fires remains subject to air quality control constraints, increased fire use may continue to be limited geographically and temporally. Despite these constraints there are still large tracts of forested land where fire use can be safely and effectively implemented. Federal and state agencies need to support further policy development to overcome the risks associated with both allowing lightning-ignited fires to burn and intentional burning. Furthermore, agencies and managers must not only reach out to educate the public on the benefits of burning, they need to make good faith efforts to include interest groups and other members of the public into the planning of such projects.

Products

Presentations

2006, Yosemite Fire Symposium, Yosemite National Park, California.
topic: Spatial patterns of large natural fires in Sierra Nevada wilderness areas

2006, 3rd International Fire Ecology and Management Congress, San Diego, California.
topic: Managing natural fire in Sierra Nevada wilderness areas

Papers

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

Collins, B. M., and S. L. Stephens. 2007. Managing natural fire in Sierra Nevada wilderness areas. Frontiers in Ecology and the Environment 5 (*in press*).

Collins, B. M., and S. L. Stephens. 2007. Fire scarring patterns in Sierra Nevada wilderness areas burned by multiple wildland fire use fires. Fire Ecology (*in review*).

Collins, B. M., J. D. Miller, A. E. Thode, N. M. Kelly, J. W. van Wagtendonk, and S. L. Stephens. 2007. Natural wildfires become self-limiting in Sierra Nevada mixed conifer forests. Proceedings of the National Academy of Sciences (*in review*).

Website

http://nature.berkeley.edu/stephens-lab/JFSPlinks.htm

References

Agee J.K. 1998. The landscape ecology of Western forest fire regimes. Northwest Science 72: 24-34.

Battles J.J., Shlisky A.J., Barrett R.H., Heald R.C. and Allen-Diaz B.H. 2001. The effects of forest management on plant species diversity in a Sierran conifer forest. Forest Ecology and Management 146: 211-222.

Bekker M.F. and Taylor A.H. 2001. Gradient analysis of fire regimes in montane forests of the southern Cascade range, Thousand Lakes Wilderness, California, USA. Plant Ecology 155: 15-28.

Bradshaw L.S., Deeming J.E., Burgan R.E. and Cohen J.D. 1983. The 1978 National Fire Danger Rating System: technical documentationGTR-INT-169. U. S. Department of Agriculture, Forest Service, Intermountain and Range Experiment Station, Ogden, UT. p. 44.

Breiman L., Friedman J.H., Olshen R.A. and Stone C.G. 1984. Classification and Regression Trees. Wadsworth, Belmont, CA, USA.

Brewer C.K., Winne J.C., Redmond R.L., Opitz D.W. and Mangrich M.V. 2005. Classifying and mapping wildfire severity: a comparison of methods. Photogrammetric Engineering and Remote Sensing 71: 1311-1320.

Brown P.M., Kaufmann M.R. and Shepperd W.D. 1999. Long-term, landscape patterns of past fire events in a montane ponderosa pine forest of central Colorado. Landscape Ecology 14: 513-532.

Brown P.M. and Wu R. 2005. Climate and disturbance forcing of episodic tree recruitment in a southwestern ponderosa pine landscape. Ecology 86: 3030-3038.

Calbk M.E., White D. and Kiester A.R. 2002. Assessment of spatial autocorrelation in empirical models of ecology. *In* Scott J. M., Heglund P. J., Morrison M. L., Haufler J. B., Raphael M. G., Wall W. A. and Samson F. B. (eds.), Predicting Species Occurrences: Issues of Scale and Accuracy, pp. 429-440. Island Press, Washington, D.C.

Caprio A.C. and Graber D.M. 2000. Returning fire to the mountains: can we successfully restore the ecological role of pre-Euroamerican fire regimes in the Sierra Nevada? *In* Proceedings of the wilderness science in a time of change conference, pp. 1-12. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Missoula, MT.

Collins B.M., Kelly N.M., van Wagtendonk J.W. and Stephens S.L. 2007. Spatial patterns of large natural fires in Sierra Nevada wilderness area. Landscape Ecology 22: 545-557.

Collins B.M. and Stephens S.L. 2007. Managing natural wildfires in Sierra Nevada wilderness areas. Frontiers in Ecology and the Environment in press.

Cook E.R., Meko D.M., Stahle D.W. and Cleaveland M.K. 1999. Drought reconstructions for the continental United States. Journal of Climate 12: 1145-1162.

Crosby J.S. and Chandler C.C. 2004. Get the most from your windspeed observation. Fire Management Today 64: 53-55.

De'ath G. and Fabricius K.E. 2000. Classification and regression trees: a powerful yet simple technique for ecological data analysis. Ecology 81: 3178-3192.

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

ESRI 2004. ArcGIS 9.0. Copyright ©2004 ESRI Inc., Redlands, California.

Finney M.A., McHugh C.W. and Grenfell I.C. 2005. Stand- and landscape-level effects of prescribed burning on two Arizona wildfires. Canadian Journal of Forest Research 35: 1714-1722.

Fule P.Z., Crouse J.E., Heinlein T.A., Moore M.M., Covington W.W. and Verkamp G. 2003. Mixed-severity fire regime in a high-elevation forest of Grand Canyon, Arizona, USA. Landscape Ecology 18: 465-485.

Fule P.Z. and Laughlin D.C. 2007. Wildland fire effects on forest structure over an altitudinal gradient, Grand Canyon National Park, USA. Journal of Applied Ecology 44: 136-146.

Hessburg P.F., Agee J.K. and Franklin J.F. 2005. Dry forests and wildland fires of the inland Northwest USA: contrasting the landscape ecology of the pre-settlement and modem eras. Forest Ecology and Management 211: 117-139.

Hosmer D.W. and Lemesow S. 2000. Applied logistic regression, 2nd edition. Wiley, New York.

Key C.H. and Benson N.C. 2005. Landscape assessment: ground measure of severity, the composite burn index, and remote sensing of severity, the normalized burn ratioGTR-RMRS -164-CD. U. S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Kitzberger T., Brown P.M., Heyerdahl E.K., Swetnam T.W. and Veblen T.T. 2007. Contingent Pacific-Atlantic Ocean influence on multicentury wildfire synchrony over western North America. Proceedings of the National Academy of Sciences of the United States of America 104: 543-548.

McGarigal K., Cushman S.A., Neel M.C. and Ene E. 2002. FRAGSTATS: spatial pattern analysis program for categorical maps. Computer software program produced by the authors at the University of Massachusetts, Amherst. Available at the following web site: www.umass.edu/landeco/research/fragstats/fragstats.html.

Miller C. and Urban D.L. 2000a. Connectivity of forest fuels and surface fire regimes. Landscape Ecology 15: 145-154.

Miller C. and Urban D.L. 2000b. Modeling the effects of fire management alternatives on Sierra Nevada mixed-conifer forests. Ecological Applications 10: 85-94.

Miller J.D. and Thode A.E. 2007. Quantifying burn severity in a heterogeneous landscape with a relative version of the delta Normalized Burn Ratio (dNBR). Remote Sensing of Environment doi: 10.1016/j.rse.2006.12.006: in press.

Morgan P., Hardy C.C., Swetnam T.W., Rollins M.G. and Long D.G. 2001. Mapping fire regimes across time and space: understanding coarse and fine-scale fire patterns. International Journal of Wildland Fire 10: 329-342.

North M., Hurteau M., Fiegener R. and Barbour M. 2005. Influence of fire and El Nino on tree recruitment varies by species in Sierran mixed conifer forest. Forest Science 51: 187-197.

Rollins M.G., Swetnam T.W. and Morgan P. 2001. Evaluating a century of fire patterns in two Rocky Mountain wilderness areas using digital fire atlases. Canadian Journal of Forest Research 31: 2107-2123.

Rollins M.G., Morgan P. and Swetnam T. 2002. Landscape-scale controls over 20^{th} century fire occurrence in two large Rocky Mountain (USA) wilderness areas. Landscape Ecology 17: 539-557.

Schoennagel T., Veblen T.T. and Romme W.H. 2004. The interaction of fire, fuels, and climate across Rocky Mountain forests. Bioscience 54: 661-676.

Stephens S.L. 2004. Fuel loads, snag abundance, and snag recruitment in an unmanaged Jeffrey pine-mixed conifer forest in Northwestern Mexico. Forest Ecology and Management 199:

103-113.
Stephens S.L. and Fry D.L. 2005. Spatial distribution of regeneration patches in an old-growth Pinus jeffreyi-mixed conifer forest in northwestern Mexico. Journal of Vegetation Science 16: 693-702.
Stephens S.L. and Gill S.J. 2005. Forest structure and mortality in an old-growth Jeffrey pine-mixed conifer forest in north-western Mexico. Forest Ecology and Management 205: 15-28.
Stephenson N.L. 1999. Reference conditions for giant sequoia forest restoration: structure, process, and precision. Ecological Applications 9: 1253-1265.
Thode A.E. 2005. Quantifying the fire regime attributes of severity and spatial complexity using field and imagery data. PhD dissertation, University of California, Davis.
Van Horne M.L. and Fule P.Z. 2006. Comparing methods of reconstructing fire history using fire scars in a southwestern United States ponderosa pine forest. Canadian Journal of Forest Research 36: 855-867.
van Wagtendonk J.W. 1995. Large fires in wilderness areas. *In* Proceedings of the symposium on fire in wilderness and park management, pp. 113-116. U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Missoula, MT.
van Wagtendonk J.W., Root R.R. and Key C.H. 2004. Comparison of AVIRIS and Landsat ETM+ detection capabilities for burn severity. Remote Sensing of Environment 92: 397-408.
Westerling A.L., Hidalgo H.G., Cayan D.R. and Swetnam T.W. 2006. Warming and earlier spring increase western US forest wildfire activity. Science 313: 940-943.