Ecosystems (2009) 12: 114–128
DOI: 10.1007/s10021-008-9211-7

ECOSYSTEMS
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Interactions Among Wildland Fires in a Long-Established Sierra Nevada Natural Fire Area

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Abstract

We investigate interactions between successive naturally occurring fires, and assess to what extent the environments in which fires burn influence these interactions. Using mapped fire perimeters and satellite-based estimates of post-fire effects (referred to hereafter as fire severity) for 19 fires burning relatively freely over a 31-year period, we demonstrate that fire as a landscape process can exhibit self-limiting characteristics in an upper elevation Sierra Nevada mixed conifer forest. We use the term 'self-limiting' to refer to recurring fire as a process over time (that is, fire regime) consuming fuel and ultimately constraining the spatial extent and lessening fire-induced effects of subsequent fires. When the amount of time between successive adjacent fires is under 9 years, and when fire weather is not extreme (burning index <34.9), the probability of the latter fire burning into the previous fire area is extremely low. Analysis of fire severity data by 10-year periods revealed a fair degree of stability in the proportion of area burned among fire severity classes (unchanged, low, moderate, high). This is in contrast to a recent study demonstrating increasing high-severity burning throughout the Sierra Nevada from 1984 to 2006, which suggests freely burning fires over time in upper elevation Sierra Nevada mixed conifer forests can regulate fire-induced effects across the landscape. This information can help managers better anticipate short- and long-term effects of allowing naturally ignited fires to burn, and ultimately, improve their ability to implement Wildland Fire Use programs in similar forest types.

Key words: Wildland fire use; WFU; Prescribed natural fire; Fire management; Fire ecology; Fire severity; Self-limiting; Fire policy.

Introduction

Throughout much of the western U.S. wildfires have been increasing in extent and frequency over the last several decades (Stephens 2005; Westerling and others 2006; Holden and others 2007). Additionally, a recent study has demonstrated a regional increase in the severity of fires for much of the forested land in California and western Nevada

Received 29 April 2008; accepted 22 September 2008; published online 4 November 2008

BC wrote paper, performed analysis; JM gathered/processed data, performed analysis, contributed to writing; AT gathered/processed data, conducted field research; MK contributed new methods for analysis; JvW performed analysis, conceived the study; SS designed study, contributed to writing.

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(Miller and others 2008). Concerns over these increases have raised a suite of questions aimed at assessing the historical role of fire in forested ecosystems (Brown and others 1999; Stephenson 1999; Heyerdahl and others 2001; Taylor and Skinner 2003; Schoennagel and others 2004; Brown and Wu 2005; Collins and Stephens 2007; Millar and others 2007). The consensus among these studies of historical fire occurrence is that in many drier, mid- to low-elevation forest types throughout the western U.S., fire was a critical process that regulated forest structure and maintained relatively low fuel loads (Stephens and others 2008). Fire exclusion policies have effectively removed this process and altered forest structure and fuel loads, such that many of these forest types are prone to uncharacteristically severe and extensive fire (Parsons and Debenedetti 1979; Allen and others 2002; Schoennagel and others 2004; Hessburg and others 2005; Donovan and Brown 2007; Stephens and others 2007).

Although structural changes in these drier, mid- to low-elevation forest types are thought to be contributing to increased wildfire activity (Donovan and Brown 2007), these changes may not be singularly responsible for the current wildfire problem. Several previous studies have established robust, regional-scale linkages between climatic conditions and wildland fire activity throughout the western U.S., from interannual time scales (Simard and others 1985; Swetnam and Betancourt 1990; Westerling and Swetnam 2003; Collins and others 2006), to decadal scales (Kitzberger and others 2007), up to centennial scales (Pierce and others 2004). Additionally, recent increases in surface temperatures and earlier onset of spring snowmelt have been connected with observed increases in large wildfire frequency (Westerling and others 2006). These increases in wildfire activity are expected to continue as the general warming trend in the long-term Atlantic Ocean—western U.S. climate teleconnection (Atlantic Multidecadal Oscillation) appears imminent (McCabe and others 2004; Kitzberger and others 2007). The combined effects of anticipated changes in regional climate and the observed changes in forest structure within many western U.S. forest types present a daunting potential for damage to some forested ecosystems and human communities. Fire policy will need to adapt to these changing conditions (McKenzie and others 2004; Stephens and Ruth 2005; Moritz and Stephens 2008).

One of the strategies for reducing potential wildfire activity is to mechanically manipulate fuel (live and dead biomass) in forests such that the intensity of subsequent fires is reduced, thus alleviating the negative effects of uncharacteristically severe fire on some ecosystems (Agee and Skinner 2005). However, high costs associated with implementing some fuel reduction projects limit the applicability across many forested landscapes (Hartsough and others 2008). Furthermore, the mechanical disturbance of cutting and removing fuels is controversial in some areas, and infeasible in more remote wilderness areas. An alternative approach is to use fire as a tool for reducing fuel loads with the assumption that fuel consumed by these fires may limit the extent and/or effects of subsequent fires. This strategy is most effective in drier forest types historically associated with frequent fire (Agee and Skinner 2005). Prescribed fire treatments have been shown to both constrain the extent and reduce the fire-induced effects of multiple large, western U.S. wildfires (Finney and others 2003; Finney and others 2005). However, very little is known on whether wildland fires, if allowed to burn freely under a range of environmental conditions, can limit the extent and effects of subsequent fires over a landscape (but see van Wagtendonk 2004; Bigler and others 2005).

In this study, we ask if fire as a natural process in a mixed conifer forested landscape can be 'self-limiting'? We use the term 'self-limiting' to refer to recurring fire as a process over time (that is, fire regime) consuming fuel and ultimately constraining the spatial extent and lessening fire-induced effects of subsequent fires. We hypothesize that shorter time intervals between adjacent fires will result in the latter fire being either constrained in extent or reburning an area burned in the previous fire under lessened fire severity (Figure 1). If intervals between adjacent fires are longer we expect no spatial constraining effect of the previous fire on the latter fire and little or no effect on severity within the reburned area. However, we also hypothesize that fire weather may substantially influence these interactions. More extreme fire weather may allow for areas to reburn even if intervals between fires are short or may lead to greater fire severity (Figure 1) (Finney and others 2003). If fires are constrained in extent and/or effects by previous fires, then increased use of wildland fires can be a viable and economically feasible solution for mitigating potential increases in uncharacteristically large and severe wildfires (Ingalsbee 2001; Dale and others 2005). If not, intentionally burning, or allowing wildlands to burn naturally may be an ineffective strategy for reducing the extent and severity of subsequent wildfires.

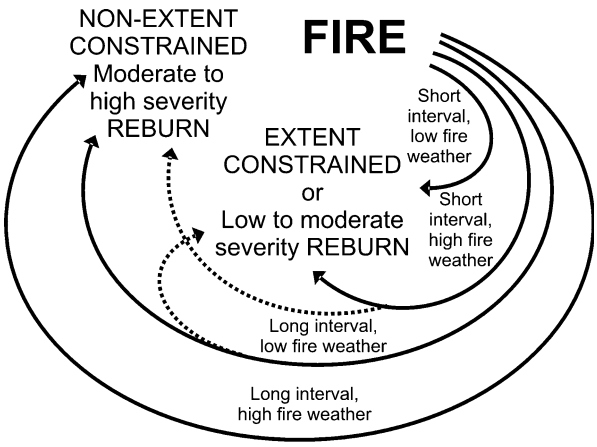

Figure 1. Interactions between successive fires resulting in the latter fire: (1) being constrained in spatial extent by the previous fire or lessened fire severity if the previously burned area is reburned, (2) no spatial constraint of the previous fire on either the extent or severity of the latter fire. We propose the dominant factors controlling these interactions are the time between successive fires (interval) and fire weather during the latter fire. *Solid lines* are primary expected pathways and *dashed lines* are secondary pathways.

Due to the widespread and long-standing practice of fire suppression there are only a few areas throughout the western U.S., where interactions among multiple, naturally occurring fires can be studied (see Rollins and others 2002; Collins and others 2007; Holden and others 2007). The Illilouette Creek basin in Yosemite National Park, USA, is one such area. For over 30 years this remote wilderness area has employed a policy (now called Wildland Fire Use—WFU) of allowing naturally ignited fires to burn relatively unimpeded over the landscape with the intent of restoring fire as a critical ecosystem process (Parsons and others 1986). Nineteen fires (>40 ha) burned within the Illilouette Creek basin since the beginning of the WFU program in 1972, providing the relatively rare opportunity to identify patterns that may explain interactions among multiple overlapping and adjacent, actual fires (as opposed to simulated fires). Our goal is to identify the conditions of fuels, weather, and/or topography that influence whether a wildland fire is constrained in extent by a previous fire or burns over a previously burned area. Additionally, we investigate potential trends in post-fire effects on the dominant vegetation, or fire severity, for these naturally occurring fires over the last 30 years. We analyzed mapped fire perimeters (Morgan and others 2001), along with recently developed satellite-based estimates of fire severity (Thode 2005; Miller and Thode 2007), to characterize the attributes of, and identify interactions among, the 19 WFU fires that burned between 1972 and 2006 in the Illilouette Creek basin.

Methods

Study Area

Illilouette Creek basin is in the central Sierra Nevada, California, USA (Figure 2). The basin is over 15,000 ha with elevations ranging from 1400 to nearly 3000 m for the surrounding ridges. The climate is Mediterranean with cool, moist winters, and warm, generally dry summers. Average January minimum temperatures range from −2 to −5°C, whereas average July maximum temperatures range from 24 to 32°C (1992–2007, Crane Flat Lookout station). Precipitation varies with elevation and is predominantly snow, with annual averages near 100 cm. The forests in Illilouette Creek basin are dominated by Jeffrey pine (*Pinus jeffreyi*), white fir (*Abies concolor*), red fir (*Abies magnifica*), and lodgepole pine (*Pinus contorta* var. *murrayana*), and are interspersed with meadows and shrublands. The historical fire regime in the Illilouette basin predominantly consisted of frequent surface fires. Collins and Stephens (2007) reported a historical mean fire return interval of 6.3 years for the Illilouette basin (between 1700 and 1900), with a historical fire rotation (defined as the length of time necessary to burn a cumulative area equivalent to the study area—which was ~500 ha.) of 24.7 years. The same study also found that throughout the recent WFU period (1973–2005) the mean fire return interval and fire rotation did not differ substantially from estimates of these parameters derived from the historical reconstruction (6.8 and 32.9 years, respectively).

Spatial and Weather Data

We obtained a digital fire atlas consisting of all fire perimeters for which a record exists dating back to 1930 within Yosemite National Park from park personnel (van Wagtendonk 1986). For our analysis, we only examined WFU fires that occurred between 1972 and 2006 within the Illilouette Creek basin. The fire atlases are a best approximation of actual burn perimeters, but do not provide information on the spatial heterogeneity of burning within fire areas (Morgan and others 2001). We used satellite-based estimates of fire severity to characterize this heterogeneity within fire areas. Based on the availability of satellite imagery, fire severity data for fires prior to 1984 (n = 8) were

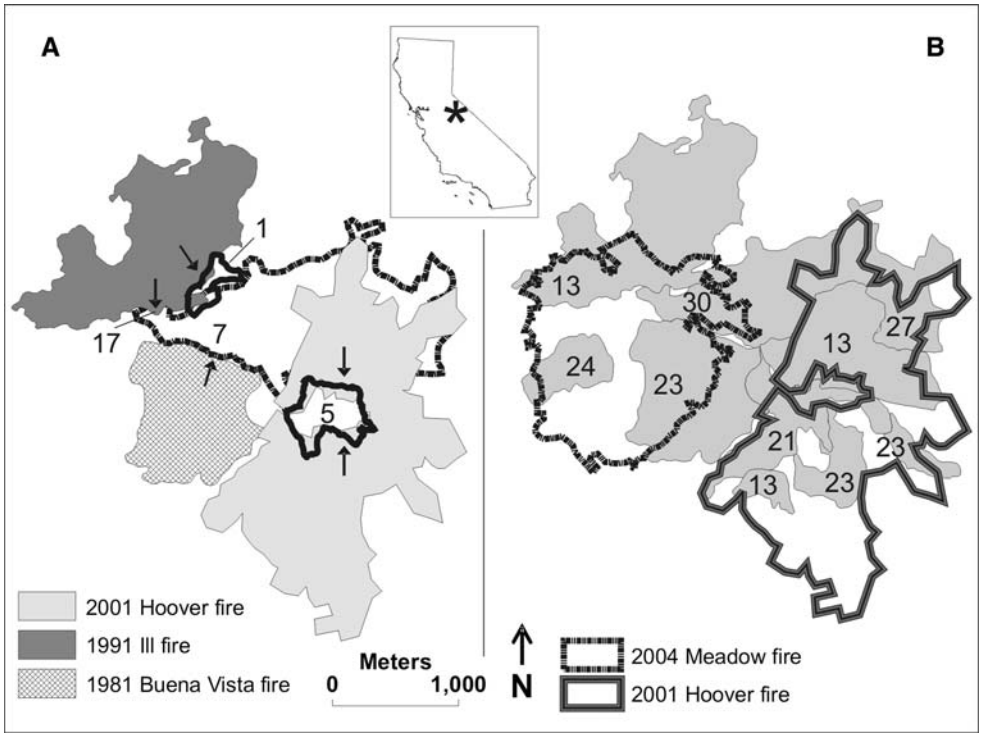

Figure 2. Examples where wildland fires were spatially constrained by previous fires (**A**) and where fires burned over area previously burned during the study period (1972–2006) (**B**) in the Illilouette Creek basin, California, USA. The *numbers* within each fire outline indicate the time (years) since each previous fire. The *arrows* (**A**) show where each listed fire was constrained in extent by a previous fire. The * depicts the research location.

derived from images obtained using a sensor (Landsat MSS) which only had four bands (as compared to seven bands on the sensor used for fires that occurred in 1984 or later) and had larger pixels (66 m, as compared to 30 m for the later sensor). As a result, fire severity data from fires that occurred between 1974 and 1983 were developed using a relative version of the difference between pre-fire and post-fire Normalized Difference Vegetation Index (RdNDVI) (Thode 2005). NDVI has often been used in studies to characterize changes in forest cover due to stand replacing disturbance (Kasischke and others 1993; Cohen and others 2002; Wilson and Sader 2002) and is defined below (equation 1).

$$\mathrm{NDVI} = \frac{(\mathrm{NIR} - \mathrm{RED})}{(\mathrm{NIR} + \mathrm{RED})}, \tag{1}$$

$$\mathrm{RdNDVI} = \frac{(\text{pre-fireNDVI} - \text{post-fireNDVI})}{\text{pre-fireNDVI}}. \tag{2}$$

For fires that occurred in 1984 or later (n = 11) we used a relative version of the differenced Normalized Burn Ratio (RdNBR), which incorporates short-wave infrared wavelengths that improve sensitivity to ash, char, and leaf water content (Key and Benson 2005a; Kokaly and others 2007; Miller and Thode 2007). NBR is computed similar to NDVI except Landsat TM short-wave infrared band 7 is used instead of the red band (Lopez Garcia and Caselles 1991; Key and Benson 2005a) and is defined below (equation 3).

$$\mathrm{NBR} = \frac{(\mathrm{NIR} - \mathrm{SWIR})}{(\mathrm{NIR} + \mathrm{SWIR})}, \tag{3}$$

$$\mathrm{RdNBR} = \frac{(\text{pre-fireNBR} - \text{post-fireNBR})}{\sqrt{(\mathrm{ABS}(\text{pre-fireNBR}))}}. \tag{4}$$

Geometric co-registration, atmospheric conditions, topography, surface moisture, seasonal phenology, and solar zenith angle all influence analysis of multi-temporal data in change detection protocols (Singh 1989; Coppin and Bauer 1996). All images used in this project were geometrically registered using terrain correction algorithms (Level 1T) by the EROS Data Center. All images were subset to within a few kilometers of the perimeter for each fire before analysis and were shifted when necessary to ensure registration with one pixel. Pre- and post-fire image dates were matched within 1-month anniversary dates to minimize differences in phenology and sun-angle. All image dates ranged between the first week of July and the first week of September to maintain high sun elevation angles. All post-fire images were acquired during the summer 1 year after the fire to correspond to the time field data were collected to calibrate the indices to a ground measured indicator of severity (Key and Benson

2005a, b). Field data collection, as well as calibration and validation of field data, are explained in Miller and Thode (2007) and Miller and others (2008).

We converted all images to reflectance to correct for sun-sensor geometry (Markham and Barker 1986; Chandler and Markham 2003). For NDVI images, we employed a dark-object subtraction as a first-order correction for atmospheric scattering (Avery and Berlin 1992). For NBR images, we did no atmospheric correction because NBR employs only near and short-wave infrared wavelengths that are minimally affected by atmospheric scattering (Avery and Berlin 1992). During the summer months in our study area, which has a Mediterranean climate, radiometric errors are minimal relative to the change in reflectance due to stand replacing deforestation (Cohen and others 1998). Satellite values were not corrected for topographic shading because NDVI and NBR are ratios and topographic effects cancel when atmospheric scattering is minimal (Kowalik and others 1983; Ekstrand 1996). Additionally, to account for potential differences in phenology and surface moisture conditions we subtracted the mean value (dNDVI or dNBR) of an unchanged area outside the fire perimeter from the dNDVI or dNBR images. This normalization procedure ensures unburned areas have values around 0.

Thresholding differenced images in a change detection protocol can lead to misclassification in areas with heterogeneous vegetation. Stratification by vegetation type is a method that has been successfully used to create homogeneous landscapes out of heterogeneous ones prior to classification (Strahler 1981; Ekstrand 1994; Salvador and others 2000; Miller and Yool 2002). We eliminate the requirement for stratification through normalizing the differenced NDVI and NBR indices by dividing by a function of the pre-fire values to account for the pre-fire vegetation condition. One set of thresholds can then be used for estimating fire severity classes across multiple fires and vegetation types, at least in the same geographic area (Zhu and others 2006; Miller and Thode 2007). Because we normalize the dNDVI and dNBR images so that pixel values around 0 represent the unburned condition, pixels that see a decrease in vegetation cover theoretically range between 0 for unburned areas and some maximum value representing 100% vegetation mortality. In practice, RdNDVI and RdNBR continue to vary in value after complete vegetation mortality has occurred because the NDVI and NBR are sensitive to ash, char, soil moisture, and substrate composition (Bannari and others 1996; Chuvieco and others 2006; Kokaly and others 2007).

Thresholds for NDVI and NBR images were developed using Landsat TM. The NDVI thresholds were applied to the MSS-derived RdNDVI data because we did not have historic field data to calibrate MSS images. Our use of relativized indices allows extrapolation of TM-derived thresholds to MSS images even though MSS and TM red and near-infrared bands used to compute NDVI are not identical and the MSS sensor data are recorded at a coarser radiometric resolution. The coarser radiometric resolution of the MSS data only affects the precision (that is, variance) and not the accuracy of the reflectance data. MSS-derived NDVI values will be lower than those derived from TM due to different center wavelengths and bandwidths (760–900 nm for TM vs. 800–1100 nm for MSS) (Jensen 1996; Teillet and others 1997). Simulation studies have shown that differences between MSS and TM-derived NDVI can be modeled with linear regression to within a precision of 1–2% (Steven and others 2003). Because the RdNDVI index is computed as the ratio of dNDVI to the pre-fire NDVI value (2), linear differences in NDVI values for different sensors cancel so long as the same sensor is used for paired pre- and post-fire images and the sensor is stable. Sensor stability in Landsat 1 MSS may be questionable given that reflectance values changed on the order of 25–32% over the life cycle of the sensor. It is hypothesized that this range in reflectance values may be due to deterioration of internal optics (Horan and others 1974; Nelson 1985). We therefore used image-to-image regression of unchanged pixels as relative radiometric normalization for fires included in this study that used Landsat 1 image pairs (Yuan and Elvidge 1996). All other fires in this study mapped with the MSS sensor were acquired by Landsat 2 or 3 whose reflectance values were relatively stable, at least through 1978 (Markham and Barker 1986). There has been no evaluation of data acquired by Landsat 2 or 3 after 1978. All but one fire included in this study used the same sensor for both pre- and post-fire images and any small temporal drift in radiometric properties would result in errors much smaller than the change in reflectance values due to a stand-replacing event (Cohen and others 1998). In the absence of existing historical reference data for validation, several studies have demonstrated effective use of visual interpretation to validate classifications of stand replacing events (Hall and others 1984; Cohen and others 1998; Hayes and Sader 2001). We therefore visually checked severity classifications against pre- and

post-fire reflectance images for each fire to verify classifications were reasonable and aligned with patterns of tree mortality. The relative index methodology we used and the manner in which we normalized the dNDVI data so that unburned areas had values around 0 appeared to substantially correct for MSS calibration issues.

Miller and Thode (2007) reported accuracies for RdNBR-derived classifications of severity on a combination of forested and shrub plots as measured with the Composite Burned Index (CBI) (Table 1) (Key and Benson 2005b). Accuracies derived from RdNDVI classifications using the same CBI field plots were very similar (Table 2) with the exception of the unchanged severity category. CBI field data independently collected by USGS and NPS personnel in seven fires in Yosemite NP, reported on in Zhu and others (2006) demonstrate similar accuracies to those reported in Miller and Thode (2007). One of these seven Yosemite fires (2001 Hoover fire) occurred in the Illilouette Creek basin and was included in this study. Because the CBI protocol consists entirely of ocular estimates, at least some of the errors in the confusion matrices are most likely due to variation in estimated field values given that visual estimates of cover can be biased and have high variances (Korhonen and others 2006). The unchanged burn severity class is defined as areas where 1-year post-fire was relatively indistinguishable from the pre-fire conditions, which does not always indicate the area did not burn (Miller and Thode 2007).

Using the software package ArcGIS®, we analyzed fire atlases and fire severity images to identify wildland fires that appeared to be constrained in spatial extent by a previous fire (referred to hereafter as ''extent-constrained''—Figure 2A) or burned over an area previously burned during the study period (1972–2006) (referred to hereafter as ''reburn''—Figure 2B). The rule used to define extent-constrained 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 aerial photographs to identify landscape features that would act as natural firebreaks (rock outcrops, wet meadows, and so on). Extent-constrained 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.

In addition to the fire severity images, we assembled datasets for dominant vegetation, slope gradient, and weather for each WFU fire. We obtained maps of dominant vegetation from Yosemite National Park personnel, from which we identified four different forest vegetation types (red

Table 1. RdNBR Classification Confusion Matrix for CBI Field Calibration Plots from 14 Fires in the Sierra Nevada

Class name	Unchanged	Low	Moderate	High	Total	User's accuracy (%)
Unchanged	21	27	2		50	42.0
Low	8	115	79	9	211	54.5
Moderate		61	157	49	267	58.8
High		8	55	150	213	70.4
Total	29	211	293	208	741	
Producer's accuracy (%)	72.4	54.5	53.6	72.1		59.8

Includes forested and non-forested plots, reproduced from Miller and Thode (2007).

Table 2. RdNDVI Classification of CBI Confusion Matrix for Plots from 14 Fires in the Sierra Nevada

Class name	Unchanged	Low	Moderate	High	Total	User's accuracy (%)
Unchanged	16	28	3	3	50	32.0
Low	11	130	64	10	215	60.5
Moderate	1	52	171	46	270	63.3
High		1	54	151	206	73.3
Total	28	211	292	210	741	
Producer's accuracy (%)	57.1	61.6	58.6	71.9		63.2

Includes forested and non-forested plots.

fir—lodgepole pine, red-fir—Jeffrey pine, Jeffrey pine, lodgepole pine) and one shrubland type. Rather than averaging across each fire, we subsampled the fire severity, slope gradient, and dominant vegetation for the area of overlap (reburn) or adjacency (extent-constrained). For each extent-constrained interaction, we created a 200-m buffer around the fire that appeared to be constrained, just along the area of the common border (extending only into the area that did not reburn) and extracted data for each of the spatial variables. For reburn interactions, we only extracted data that were within the area burned over. In each case, we averaged the spatial variables over the reburned area to get one value for each of the following variables for each reburn and extent-constrained interaction: fire severity in the previous fire (mean RdNBR), time since previous fire, dominant vegetation type, slope gradient, and burning index during the time of the second fire. The burning index is strongly influenced by weather, however it also incorporates topographic and fuel influences on fire spread and energy release (Bradshaw and others 1984). Because the fuel component of the burning index calculation is based on relatively coarse-scale estimates of fuel structure and abundance, or fuel model, we use only one fuel model for the entire Illilouette Creek basin (NFDRS fuel model H—short-needle, normal dead). As such, we use the burning index as a proxy for fire weather. Daily burning index values were derived from daily (13:00 LT) 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. This station is approximately 30 km from the Illilouette basin and 600 m lower in elevation. 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.

To investigate trends in fire severity throughout the period of WFU fires in the Illilouette basin, we extracted pixel values from fire severity images for all the 19 fires. We compiled the fire severity pixel values to construct histograms and calculate proportions of area burned in each fire severity class for each of the 10-year periods in which WFU fires occurred: 1974–1983, 1984–1993, and 1994–2004. The first WFU fire in the Illilouette basin occurred in 1974 and the most recent fire occurred in 2004. We included 2004 with the 1993–2003 10-year period rather than having the single 2004 fire as its own period. We chose to analyze the proportions of area burned in each fire severity class by 10-year periods because the first period (1974–1983) coincided with the timing of the coarser satellite sensor, thus a slightly different fire severity index (RdNDVI as opposed to RdNBR for 1984 or later fires), as explained previously. We additionally investigated fire severity in reburned areas. We identified reburn areas based on fire perimeters that intersected, or overlapped, by more than 200 m. A total of 16 different areas that burned in previous fires reburned in subsequent fires. For each of these 16 areas, we extracted fire severity pixel values for both initial and reburn fires, and calculated the proportion of area in each fire severity class using previously determined thresholds for classes (Miller and Thode 2007).

Statistical Analyses

We used categorical tree and logistic regression analysis as complementary techniques to explain the potential influence of predictor variables on the interactions between fires and previous burns. The categorical tree analysis provides discrete values for influential predictor variables that can be used to identify potential threshold levels for a given response (Collins and others 2007), whereas logistic regression provides a continuous relationship between predictor variables and a response variable that can be used to identify comparative trends for different levels of influential predictor variables (Hosmer and Lemesow 2000). We ran the categorical tree and logistic regression analyses in the statistical software packages R (http://www.r-project.org) and SAS®, respectively. We treated each fire—previous burn interaction as an independent observation. We submit 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 (fire severity in the previous fire, time since previous fire, dominant vegetation type, slope gradient, burning index). The categorical tree is constructed by repeatedly splitting the data into increasingly homogenous groups based on the response variable, extent-constrained (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 and others 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. To assess adequacy of the categorical tree model, we report both the apparent R^2, which is one minus the apparent error, and the relative R^2, which is one minus the cross-validated relative error (Therneau and Atkinson 1997). 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 the Hosmer and Lemesow goodness-of-fit test and the area under the receiver operating characteristic curve (ROC) to assess the adequacy of our final logistic regression model (Hosmer and Lemesow 2000). The ROC is a threshold-independent measure of model discrimination in which 0.5 indicates no discrimination, 0.7–0.8 acceptable discrimination, and 0.8–0.9 excellent discrimination (Hosmer and Lemesow 2000; Das and others 2008).

We analyzed trends in fire severity throughout the period of WFU fires in the Illilouette basin by comparing the proportion of area burned in each severity class across the three 10-year periods: 1974–1983, 1984–1993, and 1994–2004. We used a Chi-squared test for the homogeneity of proportions within each fire severity class, among periods. If significant, this test indicates statistical differences in one or more of the fire severity categories among the three periods analyzed (Ott and Longnecker 2001). We performed this test comparing all the three 10-year periods together, as well as for each pairwise comparison. For the analysis of reburned areas, we tested for differences in the mean proportion of area in each fire severity class between initial fires ($n = 16$) and reburn fires ($n = 16$) with four pairwise t-tests, one for each fire severity class. We used a Bonferroni correction to account for multiple contrasts. Based on $\alpha = 0.05$ the Bonferroni correction results in $\alpha = 0.0125$.

Results

The 19 fires that occurred during the WFU period in the Illilouette basin resulted in 30 separate interactions (both extent-constrained and reburn). Both categorical tree and logistic regression analyses adequately modeled the reburn and extent-constrained interactions between fires and previous burns [categorical tree $R^2 = 0.73$ (apparent), 0.39 (relative); logistic regression ROC = 0.93]. Furthermore, both techniques identified the same two predictor variables as being important in explaining whether a fire burned over a previously burned area or was constrained in spatial extent by a previous fire: time since previous fire (logistic $P = 0.07$) and the burning index (logistic $P = 0.05$), which we use as a proxy for fire weather, as explained previously. The probability of a reburn increases as time since previous fire increases, and as fire weather becomes more extreme (that is, higher wind speed, lower relative humidity, and higher air temperature) (Figure 3A). Nine years since the previous fire appears to be a threshold at which previous fires have less of a constraining effect on the extent of subsequent fires in the mixed conifer forests of the Illilouette Creek Basin

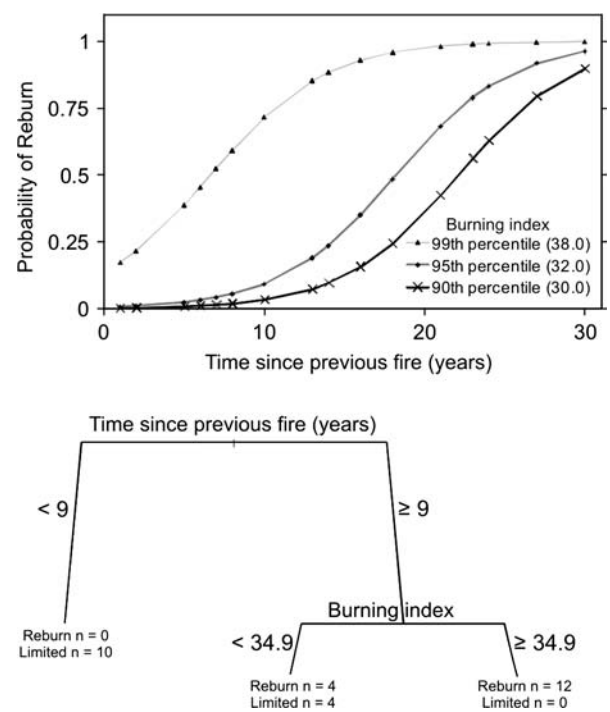

Figure 3. Logistic regression probabilities under different weather scenarios (extreme—99th, high—95th, and moderate—90th percentile burning index values) (**A**) and categorical tree break points (**B**) explaining the influence of both previous fires and weather (burning index) on extent-constrained (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. For each terminal node, we report the actual number of observations (n) of extent-constrained and reburn interactions based on the rules identified in each split. There were a total of 30 interactions among the 19 fires during the period of analysis (1972–2006).

(Figure 3B). Under 9 years, the categorical tree analysis indicates a 0 probability of reburn, and a 100% probability of an extent-constrained interaction. The results from the categorical tree analysis are particularly robust in that of the 10 interactions that occurred with a time-since-last-fire below 9 years, all 10 were actually extent-constrained (Figure 3B). In other words, there were no reburns, regardless of fire weather, when time-since-last-fire is less than 9 years. Similarly, of the 12 total interactions where time-since-last-fire was 9 or more years and a BI is 34.9 or more, all 12 resulted in a reburn (Figure 3B). The logistic regression analysis similarly demonstrates very low probabilities of reburn for high (95th percentile) and moderate (90th percentile) fire weather conditions when time since previous fire is below 9 years. However, under extreme fire weather conditions (99th percentile), the effect of time since previous fire on the probability of reburn is substantially reduced (Figure 3A). Neither model identified dominant vegetation type, slope gradient, individual weather parameters (temperature, wind, relative humidity), or the fire severity in the previous fire as influencing the interactions between fires and previous burns.

The proportion of area burned at high severity was lowest in the first 10-year period of WFU in the Illilouette basin (1974–1983) and highest in the second 10-year period (1984–1993). When examined along with the yearly area burned in each severity class (Figure 4B) it appears that earlier large WFU fires (for example, 1974, 1980, and 1981) had less high-severity burning than those that occurred in the more recent years (for example, 1991, 2001, and 2004). However, no significant difference in the proportion of area within each severity class was evident when comparing all three periods together (χ^2 $P = 0.43$) (Figure 4A). Pairwise comparisons between 1974–1983 and 1984–1993, as well as between 1974–1983 and 1994–2004 were also insignificant (χ^2 $P = 0.23$ and 0.50, respectively). Furthermore, no difference in the proportion of area in each fire severity class was evident between the two latter periods (χ^2 $P = 0.45$).

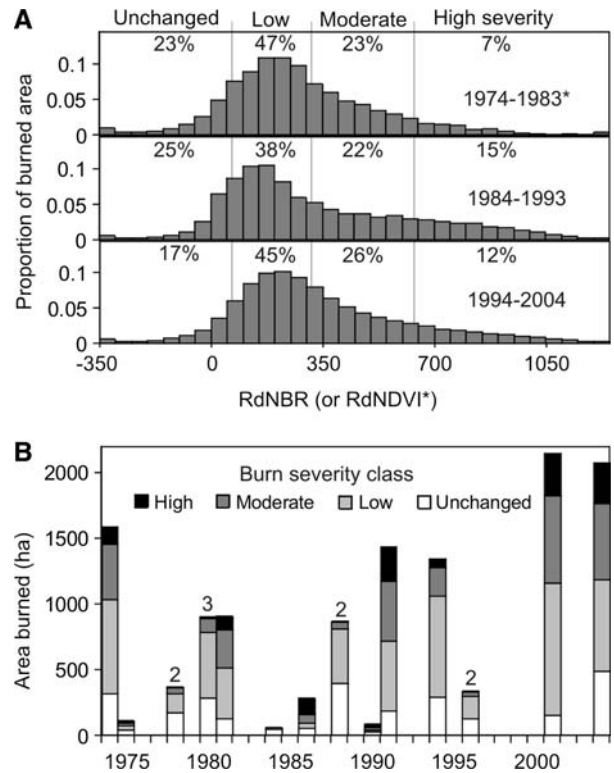

Figure 4. (**A**) Frequency distributions of fire severity pixels (RdNBR) for each 10-year period since the beginning of the natural fire program in Illilouette Creek basin, California, USA. The year 2004 was included in the previous period because it was the only fire in the current 10-year period. (**B**) Total yearly area burned differentiated by fire severity class for the period of the natural fire program within the Illilouette 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 fire severity (using the relative difference in RdNDVI) were slightly different due to differences 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.

Table 3. For Areas that Reburned, Mean Proportion of Area (Standard Deviation) by Fire Severity Class in the Initial Fire and in the Reburn Fire

	Unchanged	Low severity	Moderate severity	High severity
Initial fire	0.23 (0.13)	0.53 (0.14)	0.20 (0.13)	0.05 (0.05)
Reburn fire	0.18 (0.14)	0.42 (0.14)	0.27 (0.12)	0.12 (0.11)
P-value	0.362	0.047	0.127	0.015

P-values are from four pairwise t-tests, one for each fire severity class, in which we compared proportions in each fire severity class for initial fires (n = 16) to that in reburn fires (n = 16). We used a Bonferroni correction to account for multiple contrasts. Based on $\alpha = 0.05$ the Bonferroni correction results in $\alpha = 0.0125$.

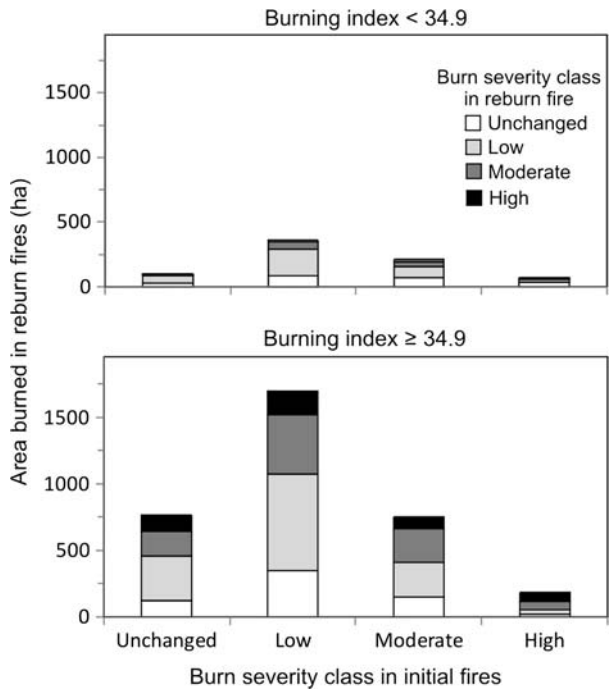

Figure 5. Total reburned area (burned twice by WFU fires) split into two groups based on the threshold value of burning index identified in the categorical tree analysis. Area burned is also differentiated by fire severity class in initial fires (*horizontal groups*) and fire severity in reburn fires (*vertical groups*).

Analysis of reburned areas indicated statistically insignificant differences in the proportion of area in the unchanged and high fire severity classes between initial and reburn fires based on a Bonferroni adjusted α-level (Table 3). The apparent increase in the proportion of high severity in reburn fires was marginally insignificant ($P = 0.015$) (Table 3). After splitting fire severity estimates for reburn areas into two groups based on the threshold value for burning index identified in the regression tree analysis, not only did the greatest proportion of total reburned area occur above the 34.9 burning index threshold, but also the greatest proportion of reburned area burned under high severity also occurred above the same threshold (Figure 5). In the less than 34.9 burning index group, high-severity burning was rare (Figure 5). A burning index of 34.9 corresponds with approximately the 97th percentile conditions for the entire period of analysis (1972–2006).

Discussion

The upper elevation mixed conifer forests in the Illilouette Creek basin historically burned frequently, at moderate- to low-intensities (Collins and Stephens 2007). Recent studies have shown that the WFU fires that occurred in the Illilouette basin over the last 33 years did not differ in extent, frequency, or effects on vegetation from reconstructions and interpretations of historical fires (Collins and others 2007; Collins and Stephens 2007). Fire regimes of this type are historically characterized as being limited by surface fuel amount and continuity (Schoennagel and others 2004; Stephens 2004; Stephens and Gill 2005). Our finding indicating the importance of time since previous fire in limiting the extent of subsequent fires supports 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 (Wright and Agee 2004).

The importance of weather in both statistical models demonstrates that fuel accumulation is not the only mechanism driving the interactions among WFU fires in upper elevation Sierra Nevada mixed conifer forests. 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 3A). The lower relative humidity, higher wind speeds, and elevated air temperatures associated with these conditions make more fuel available for burning by desiccating fuels that would not readily burn under less extreme conditions. In addition, more extreme weather leads to greater fire intensity, or increased rate of heat release, which augments fuel desiccation by preheating fuels ahead of the flaming fire front. Our findings suggest that more extreme fire weather creates conditions that may overwhelm the mechanism of fuel accumulation in limiting reburns in these forests. It is important to note that the conditions modeled under the extreme fire weather scenario are the 99th percentile fire weather; meaning only 1% of the days throughout the entire 31-year 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—previous burn interactions, fires in similar forest types will be constrained in extent by previous fires that burned relatively recently (<9 years).

We suspect that the rules governing self-limiting interactions vary among forest types. This variability is likely driven by differences in both fuel

type (that is, long- vs. short-needle and presence of herbaceous understory) and site productivity (Heyerdahl and others 2001). In the dry ponderosa pine types historically found in northern Arizona and in eastern Oregon and Washington, the long-needle litter combined with productive herbaceous understrories allowed for rapid fine fuel accumulation following fire. As Allen (2007) explained, in these forests surface fuels were often near thresholds of continuity for sustaining widespread fire. This led to fires historically recurring at intervals as low as 2–3 years (Fulé and others 1997; Heyerdahl and others 2001). In these forest types, we would expect a shorter time-since-previous-fire threshold for self-limiting behavior among adjacent fires. This short-lived effect of fire removing fuels and affecting behavior of subsequent fires are demonstrated by Finney and others (2005). They found that prescribed fire treatments that occurred less than 4 years before recent Arizona wildfires reduced fire severity in the wildfire, but as time-since-treatment exceeded 4 years there was little effect on wildfire fire severity (Finney and others 2005). On the other hand, in forest types with a larger component of short-needle tree species (for example, Douglas-fir—*Pseudotsuga menziesii*) and a less productive herbaceous understory, the time-since-last-fire threshold would likely be longer than the 9 years reported here.

Although not statistically significant, the lesser proportion of high-severity burning in the first period (1974–1983) of WFU in the Illilouette basin bears some attention. We would have assumed that the long fire-free period that preceded the implementation of WFU policy (Collins and Stephens 2007) resulted in fuel build up that would ultimately lead to greater fire severity in earlier WFU fires. Our findings do not support this assumption. It is necessary to point out that the upper elevation mixed conifer forests within the Illilouette basin are generally cooler and less productive than the mixed conifer forests commonly found throughout the western slope of the Sierra Nevada (Sugihara and others 2006). As a result, fuel build up during the 90-year-plus fire-free period may not have been as great as that in mid- to low-elevation mixed conifer forests. Additionally, forests of the Illilouette basin have never been harvested, unlike much of Sierra Nevada mixed conifer forests. The combined effects of the generally cooler temperatures and the reduced structural changes in Illilouette basin forests result in potentially decreased fire behavior compared to that in the more typical west-slope Sierra Nevada mixed conifer forests (for example, Safford and others 2008). We present this because we do not want our findings on fire severity trends to be uncritically applied to dissimilar forest types.

The lack of statistical significance in comparing proportions of area burned by severity class among the three 10-year periods suggests that the effects of WFU fires on dominant vegetation have remained relatively stable throughout the duration of the natural fire program. This is in contrast to a recent study by Miller and others (2008) demonstrating increasing high-severity burning throughout the Sierra Nevada forests from 1984 to 2006. The authors also demonstrate that this increasing trend in high severity fire co-occurs with rising regional temperatures and increased long-term precipitation. The stability in fire severity throughout the WFU period in the Illilouette basin suggests that relatively freely burning fires over time in upper elevation Sierra Nevada mixed conifer forests can regulate the fire-induced effects on vegetation across the landscape, despite regional increases in temperature and fire activity reported by both Miller and others (2008) and Westerling and others (2006). One important caveat accompanying this assertion is the fires burning during this WFU period cannot be viewed as a fully ''free-burning'' situation. Throughout this period suppression actions were taken on some naturally ignited fires because they exceeded maximum manageable area, violated air quality standards, or occurred during a WFU moratorium in 1989. However, there is doubt that if control actions had not been taken, additional high-severity patches would have burned, resulting in little or no change in the proportional distribution of severity (J. W. van Wagtendonk, 2008, personal communication).

Our findings that demonstrate no significant change in the proportional distribution of severity between initial and reburn fires are somewhat surprising. Although the increased mean proportion of high severity in reburns is insignificant given the Bonferroni corrected α-level (Table 3), this finding is opposite what we would have expected. However, after splitting reburned areas into groups based on the burning index threshold, it is apparent that a much larger proportion of reburned area burned under more severe fire weather conditions (Figure 5). The hotter, drier, and/or windier conditions, as indicated by the higher burning index values, could explain this shift toward increased fire severity in reburn areas (Agee 1993; Collins and others 2007), and again suggest that fire weather can overwhelm the 'self-limiting' effect of fire as a landscape process over time (Figure 1). It is important to note the variation in accuracies among the burn severity classes for both

RdNBR and RdNDVI classifications. The high-severity class has much higher accuracy than the low- and moderate-severity classes for both RdNBR and RdNDVI classifications (Tables 1 and 2). In addition, the unchanged class has much higher accuracy in the RdNBR classification than it does for the RdNDVI classification (Tables 1 and 2). Given this variation more confidence exists in comparisons between proportions of high severity than for other severity classes.

Conclusions

Our analysis of the interactions among relatively unmanaged wildland fires offers new insight into the controls operating on fire at a landscape scale. When fires were allowed to burn over the last 31 years in upper elevation mixed conifer forests, fires became 'self-limiting,' even under high fire weather conditions. The fact that our analysis did not identify fire severity in prior fires as contributing to the explanation of interactions with subsequent fires demonstrates that the burning effects of a fire are not as important as the time since previous fire. The proportion of area burned among fire severity classes (unchanged, low, moderate, high) remained relatively stable throughout the study period (1972–2006) in the Illilouette basin. These results are suggestive of a successful and effective WFU program in the Illilouette basin. Given that we examined 19 WFU fires burning across a single upper elevation Sierra Nevada mixed conifer forest, it is not clear to what extent these findings can be applied to other forest/fire regime types across the Sierra Nevada, or across the western U.S. As WFU is operationally increasing as a management alternative (Sexton 2006), opportunity to study relatively freely burning fires in different vegetation types and geographic areas is expanding. Carrying out such studies will allow managers to better anticipate short- and long-term effects of WFU fires and ultimately improve their ability to implement WFU programs.

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 in the western U.S., where WFU can be safely and effectively implemented. Increased WFU can reduce suppression costs, free-up suppression resources for more critical areas where social losses due to fire are greatest, and ultimately improve ecological integrity in fire-dependent systems (DeWilde and Chapin 2006). Federal and state policies can be developed to overcome the risk associated with allowing more lightning-ignited fires to burn. Furthermore, the public and special interest groups should be educated on the benefits of burning and be included in the planning of fire use programs.

Acknowledgements

We thank Tadashi Moody for his hard work in starting this project. We also thank Andy Amacher for the helpful discussions on our data analysis. This project was funded by the USDA/USDI Joint Fire Sciences Program. Any use of trade, product, or firm names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

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