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Trends and causes of severity, size, and number of fires in northwestern California, USA

J. D. Miller,[1,6] C. N. Skinner,[2] H. D. Safford,[3,4] E. E. Knapp,[2] and C. M. Ramirez[5]

[1]*USDA Forest Service, Pacific Southwest Region, Fire and Aviation Management, 3237 Peacekeeper Way, Suite 101, McClellan, California 95652 USA*
[2]*USDA Forest Service, Pacific Southwest Research Station, Redding, California 96002 USA*
[3]*USDA Forest Service, Pacific Southwest Region, Vallejo, California 94592 USA*
[4]*Department of Environmental Science and Policy, University of California, Davis, California 95616 USA*
[5]*USDA Forest Service, Pacific Southwest Region, Remote Sensing Laboratory, McClellan, California 95652 USA*

Abstract. Research in the last several years has indicated that fire size and frequency are on the rise in western U.S. forests. Although fire size and frequency are important, they do not necessarily scale with ecosystem effects of fire, as different ecosystems have different ecological and evolutionary relationships with fire. Our study assessed trends and patterns in fire size and frequency from 1910 to 2008 (all fires > 40 ha), and the percentage of high-severity in fires from 1987 to 2008 (all fires > 400 ha) on the four national forests of northwestern California. During 1910–2008, mean and maximum fire size and total annual area burned increased, but we found no temporal trend in the percentage of high-severity fire during 1987–2008. The time series of severity data was strongly influenced by four years with region-wide lightning events that burned huge areas at primarily low–moderate severity. Regional fire rotation reached a high of 974 years in 1984 and fell to 95 years by 2008. The percentage of high-severity fire in conifer-dominated forests was generally higher in areas dominated by smaller-diameter trees than in areas with larger-diameter trees. For Douglas-fir forests, the percentage of high-severity fire did not differ significantly between areas that re-burned and areas that only burned once (10% vs. 9%) when re-burned within 30 years. Percentage of high-severity fire decreased to 5% when intervals between first and second fires were >30 years. In contrast, in both mixed-conifer and fir/high-elevation conifer forests, the percentage of high-severity fire was less when re-burned within 30 years compared to first-time burned (12% vs. 16% for mixed conifer; 11% vs. 19% for fir/high-elevation conifer). Additionally, the percentage of high-severity fire did not differ whether the re-burn interval was less than or greater than 30 years. Years with larger fires and greatest area burned were produced by region-wide lightning events, and characterized by less winter and spring precipitation than years dominated by smaller human-ignited fires. Overall percentage of high-severity fire was generally less in years characterized by these region-wide lightning events. Our results suggest that, under certain conditions, wildfires could be more extensively used to achieve ecological and management objectives in northwestern California.

Key words: California; fire severity; forest fires; Klamath Mountains; relative differenced normalized burn ratio.

Introduction

The frequency of large wildfires and the annual area burned by wildfires in the western United States have both increased strongly over the last several decades (Arno and Allison-Bunnell 2002, Stephens 2005, Westerling et al. 2006, Miller et al. 2009*b*). These rising trends occur in spite of a massive fire suppression apparatus that has reduced the overall area burned by wildfires to levels that are substantially below those that occurred before the beginning of the 20th century (McKelvey et al. 1996, Sugihara et al. 2006, Stephens et al. 2007). The success of fire suppression has, ironically, fostered changes in the composition and structure of many ecosystems that are among factors believed to contribute to the current increases in burned area (Biswell 1989, Agee and Skinner 2005, Arno and Fiedler 2005, Husari et al. 2006). At the same time, changing climates are also understood to play a major part in increased fire activity and area burned (Miller 2003, McKenzie et al. 2004, Westerling et al. 2006, Miller et al. 2009*b*). Indeed, multiple lines of historical and contemporary evidence tell us that over the long term, changes in fire activity can primarily be explained by broadscale changes in climate, moderated by local changes in vegetation, fuel conditions, and human activities (Power et al. 2008, Whitlock et al. 2008, Bowman et al. 2009, Marlon et al. 2009).

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[6] E-mail: jaymiller@fs.fed.us

In the western United States, interannual variation in area burned is strongly related to patterns in temperature and precipitation, with big fire years more likely to occur in years with warm, dry conditions (Westerling et al. 2006). Factors that help generate years of low precipitation and high fire season temperatures are diverse and vary from region to region (Collins et al. 2006, Trouet et al. 2006, 2010, Westerling et al. 2006, Littell et al. 2009). Especially in regions of pronounced annual summer drought, the warming induced by climate change will likely increase the duration of the fire season and the drying of fuels, thus increasing the potential for ignitions and higher fire intensity (Swetnam 1993, Chang 1999, Williams et al. 2001, Pausas 2004).

Human impacts on fire regimes also vary from region to region, and interact with climate and ecosystem type. Humans have influence on both the occurrence of fire, through ignitions, and the behavior of fire, through changed fuel conditions and altered climates. Anthropogenic alterations of fuel conditions include grazing, logging, fire, and fire suppression, with the order of their importance varying, depending on geographic location, vegetation, environmental conditions, and site history (Agee 1993, Schoennagel et al. 2004, Sugihara et al. 2006, Noss et al. 2006).

Most studies examining trends in fire statistics have focused on fire occurrence and area rather than on the potential ecosystem effects of fire (e.g., Collins et al. 2006, Trouet et al. 2006, Westerling et al. 2006). While trends in fire size and annual area burned are compelling and interesting, they provide an incomplete picture of the potential ecosystem effects of changing fire regimes. Impacts of fires on resources such as watersheds, wildlife habitat, soils, vegetation, and forest products are better explained by the intensity of fire and its ecosystem effects, which are measured as "fire severity," than by the simple occurrence of fire or its extent (Agee 1993, Bond and van Wilgen 1996, Sugihara et al. 2006). Since many western North American ecosystems were shaped by and are adapted to frequent fire, recent increases in fire frequency and area in these ecosystems may actually be ecologically positive, but only if the severity of fire remains within bounds that maintain critical ecological processes (Collins et al. 2009).

Standardized fire severity data for a comprehensive set of historical fires across the United States have only recently become available, and to this point only a few studies have assessed fire severity levels over time in western North American landscapes (e.g., Lutz et al. 2009, Miller et al. 2009*b*, Holden et al. 2010). One study in California that quantified trends in high-severity fire was conducted in the Sierra Nevada and southern Cascades, found that annual mean and maximum fire size, burned area, and the percentage of fire area burning at high severity in mixed-conifer forests all increased between 1984 and 2006 (Miller et al. 2009*b*). Miller et al. (2009*b*) and Lutz et al. (2009), the latter in a study conducted in Yosemite National Park, both found that the proportion of area burned at higher severities increased with annual area burned. A major question is whether these results from the Mediterranean-climate zone of North America extend to other forested landscapes within and beyond the same climate zone.

The Klamath Mountains and northern Coast Ranges of northwest California are found in the most mesic part of the North American mediterranean-climate zone, and constitute a transitional area to the more humid, maritime climate of the Pacific Northwest. Northwestern California is an area of exceptional floristic diversity, due to a diverse environmental matrix born from the intersections of complex geology, rugged topography, strong climatic gradients, and biogeography (Sawyer 2007). Though the region is generally described as having had mixed-severity fire regimes strongly influenced by topography, the high frequency of fires before the 20th century led to fire effects being mostly low to moderate severity in most conifer and hardwood vegetation types (Skinner et al. 2006). Like much of the western United States, northwestern California has experienced many decades of fire suppression and other management activities that have altered the vegetation composition and structure of many areas (Taylor and Skinner 1998, 2003, Leonzo and Keyes 2010). A common assumption is that these changes in vegetation (e.g., increased forest density, increased surface and ladder fuels) will contribute to increased severity, but some researchers have challenged the premise that fuel abundance is synonymous with fuel availability for consumption (e.g., Odion et al. 2004, 2010). Assumptions that may be used by managers to make decisions may or may not be appropriate without a full understanding of how the forests of this region are responding to past activities and their interactions with fire and climate. Northwestern California is a biogeographic and climatic transition zone, and very strong environmental gradients further complicate the picture. It is, therefore, unclear to what extent the region may mimic patterns in other parts of the West, or even other parts of the world with similar climates.

In this contribution, we report results from a broadscale assessment of patterns in the extent of high-severity (forest stand-replacing) fire in a 2.35 million ha area of northern California, including the Klamath Mountains, northern California Coast Ranges, and portions of the southern Cascade Range. Previous studies of severity patterns for northwestern California have been limited to a small sample of fires from one or two years (e.g., Weatherspoon and Skinner 1995, Odion et al. 2004, Alexander et al. 2006, Thompson et al. 2007). In contrast, our study assesses temporal patterns of high-severity fire effects using a census of all fires >400 ha occurring on the four national forests of northwestern California for the period 1987–2008. A total of 650 000 ha (87% due to lightning ignitions) burned on national forest lands during the period. We stratified our data by forest type, and measured temporal trends in severity and heterogeneity ("patchiness") of high-severity fire across the study period. Even though fire severity is a more ecologically informative statistic, the period of availability for these data (1987–2008) is relatively short and may miss important trends. We therefore also used a fire perimeter data set of fires >40 ha over a longer period (1910–2008) to evaluate longer term trends in fire number, size, and annual burned area. Finally, we assessed the role of a suite of macroclimatic variables in explaining trends in both data sets.

Methods

Study area and time period

The study region is formed by the four national forests of northwestern California (Fig. 1). Geographically, the region includes the northern California Coast Range and the Klamath Mountains, as well as a portion of the southern Cascade Range. Climate is Mediterranean, with warm, dry summers and cool, wet winters; almost all precipitation falls between October and April. Strong precipitation and temperature gradients characterize the region, driven by steep, complex terrain and proximity to the Pacific Ocean (Skinner et al. 2006). Precipitation ranges from >3000 mm/yr to <500 mm/yr in the larger valleys (Skinner et al. 2006, Sawyer 2007). Elevations within fires analyzed for this study range from 43 m to 2462 m above sea level (asl).

Severity mapping

For our severity analyses, we mapped fires that occurred at least partially on USDA Forest Service (USFS) managed lands from 1987 to 2008, independent of the place of fire origin. We included in our analysis only the portions of the fires that occurred on lands administered by the four national forests in northwestern California.

Imagery used to develop vegetation burn severity (hereafter "fire severity") maps was supplied by the Monitoring Trends in Burn Severity (MTBS) program (Eidenshink et al. 2007; *available online*),[7] which maps severity of all fires > 400 ha in the western United States. MTBS maps fires using the normalized burn ratio (NBR), calculated from Landsat-TM satellite imagery, which became available in 1984 (Key and Benson 2005*a*). MTBS provides categorical severity maps derived from pre- and postfire differenced NBR (dNBR), as well as continuous dNBR and relativized dNBR (RdNBR) data (Key and Benson 2005*a*, Miller and Thode 2007). To permit inter-fire comparisons of severity, we used the RdNBR data, which removes biasing by prefire conditions (Miller and Thode 2007). A 3×3 focal mean filter was applied to the RdNBR data to minimize pixilation and to match the scale of the 90-m plots used for developing the classification thresholds (Schowengerdt 1997, Miller and Thode 2007). We categorized the RdNBR data into four levels of severity (unchanged, low, moderate, and high), based on calibrations we previously derived of RdNBR to the plot-level Composite Burn Index (CBI) severity measure (Key and Benson 2005*b*, Miller et al. 2009*a*). After classification, the raster data were converted to polygons using standard ArcGIS procedures so that contiguous classified pixels were clumped into uniform polygons.

The imagery on which our severity data are based is primarily sensitive to changes in chlorophyll, and therefore relates most directly to mortality rather than to fire intensity or effects on soils (Miller and Thode 2007). Since the imagery were all acquired the first summer after each fire, fire effects recorded by the imagery include not only immediate effects, but ecosystem responses such as mortality or resprouting that occurred during the first year after fire (Key 2006, Keeley 2009).

In this study we were primarily concerned with characterizing the modification of forested areas to a non-forested condition. Based upon regression analysis, the high-severity threshold that we used is approximately equal to 95% change in canopy cover ($r^2 = 0.56$, $P < 0.0001$; Miller et al. 2009*a*). U.S. Forest Service vegetation classification standards state that 10% prefire tree canopy cover is required for an area to be designated as forested (Brohman and Bryant 2005). At least 200% prefire cover would then be required for 95% change to result in >10% postfire tree canopy cover. Our severity threshold, therefore, results in data that most likely understate the amount of forest transformed to a non-forest condition.

The earliest prefire vegetation map available for our study area (see *Methods* below) was 1987 and, at the time of our study, imagery only for fires up through 2008 were available from the MTBS program. Thus, our study was constrained to investigating severity patterns in large wildfires that occurred in the national forests of northwestern California between 1987 and 2008. We analyzed severity in 132 fires, which is a complete census of fires > 400 ha for this period (Fig. 1). Note that some fires were composed of several adjacent fires ignited on the same day during multiple lightning strike events (so-called "complex" fires); we mapped complex fires as single fires to allow analysis of whole high-severity patches.

Vegetation stratification

It is problematic to use static maps of current vegetation to analyze severity by vegetation type and size class over time, because high-severity fire events can cause vegetation type change. USFS CALVEG maps are the only standardized, spatially complete, and frequently updated vegetation maps available for the study area (Franklin et al. 2000, USDA 2005). Although CALVEG is used as an existing vegetation map, it is also used as a timber management tool. The mapping methodology, therefore, calls for not removing any previously

[7] http://www.mtbs.gov/

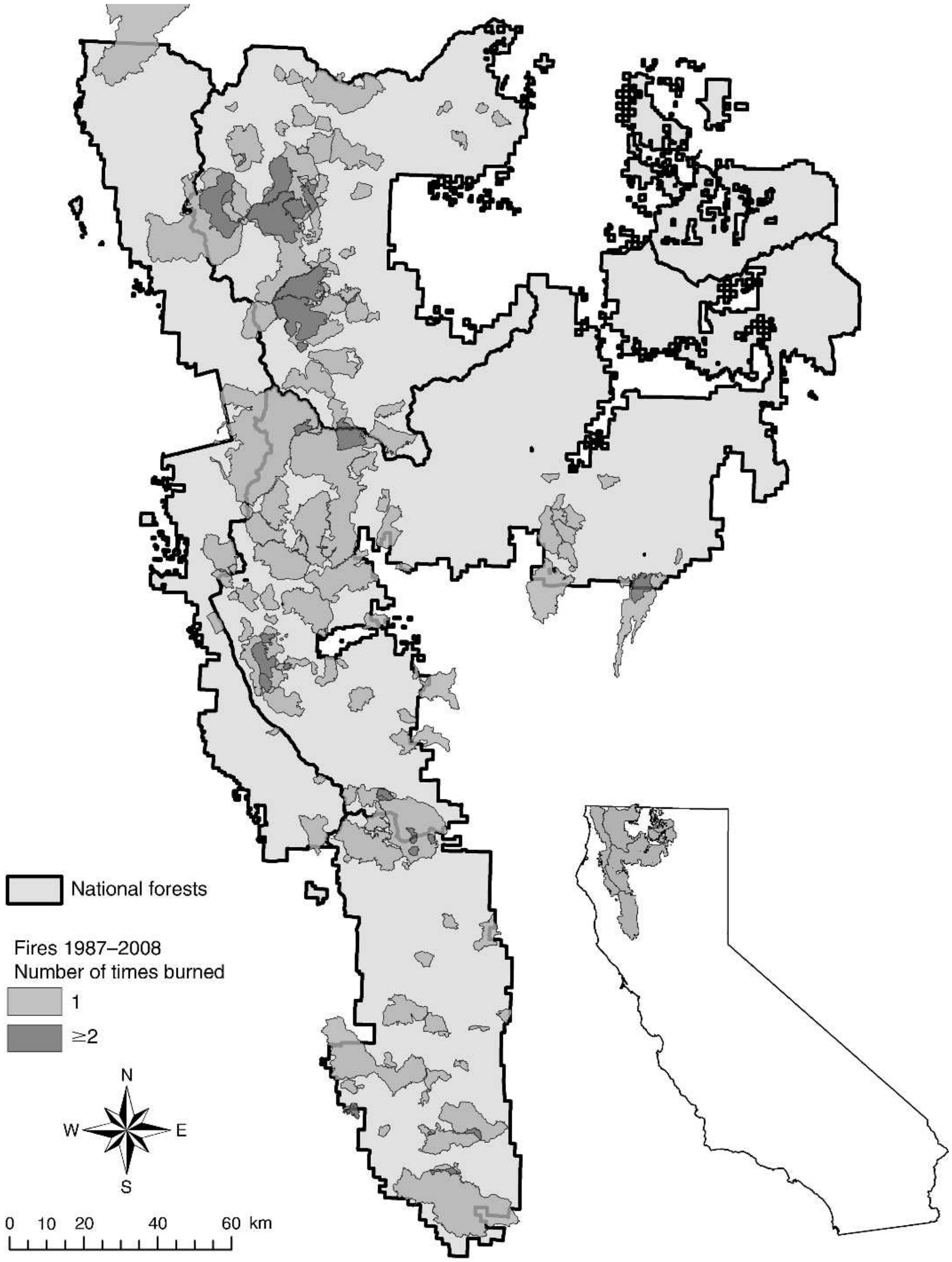

FIG. 1. Map of the overall study area. National forests of northwestern California, USA, are demarcated in light gray and outlined in black. Listed clockwise starting from the top left are the Six Rivers, Klamath, Shasta-Trinity, and Mendocino National Forests. Darker gray polygons represent 132 fires that occurred 1987–2008 for which fire severity was analyzed in the current study. Darker fire polygons represent areas that burned more than once during 1987–2008.

productive conifer forest land from the map. For example, when stand-replacing events occur in forested areas, tree density is set to zero and size to non-stocked, but the primary vegetation type is not changed. The earliest CALVEG maps with a minimum mapping unit of 1 ha, which matches the scale of our fire severity data, date from the early 1990s. However, ~25% of the MTBS fire area we analyzed burned more than once, making it impossible to use the latest CALVEG map to determine when areas were converted by fire from mature forest to early seral stages. In order to be able to stratify by prefire tree size class and density, we developed a prefire

TABLE 1. Cover and vegetation types found within the study area and the amount of each that burned (percentage, relative to total area burned) during the 1987–2008 period.

Type	Life-form	Tree canopy cover (%)	dbh (cm)	Percentage in fires 1987–2008
Cover type				
Barren (BAR)	barren	...	...	...
Conifer closed medium/large (CCM)	conifer/mixed conifer–hardwood	60–100	≥50.8	39.5
Conifer closed small (CCS)	conifer/mixed conifer–hardwood	60–100	25.4–50.8	13.1
Conifer open medium/large (COM)	conifer/mixed conifer–hardwood	10–60	≥50.8	6.3
Conifer open small (COS)	conifer/mixed conifer–hardwood	10–60	25.4–50.8	8.0
Hardwood (HDW)	hardwood	...	...	10.7
Herbaceous (HEB)	herbaceous	...	...	0.6
Sapling/poles (SAPOL)	conifer/mixed conifer–hardwood	10–100	2.54–25.4	11.5
Shrub (SHB)	shrub	...	...	9.4
Water (WAT)	water	...	...	0.0
Vegetation type		...	...	
Douglas-fir (DF)	conifer/mixed conifer–hardwood	≥10	≥25.4	32.1
Gray pine (GP)	conifer/mixed conifer–hardwood	≥10	≥25.4	0.3
Mixed conifer (MC)	conifer/mixed conifer–hardwood	≥10	≥25.4	26.6
Fir/high-elevation conifer (FIR)	conifer/mixed conifer–hardwood	≥10	≥25.4	8.6
Deciduous oak (DO)	hardwood	...	...	4.3
Live oak (LO)	hardwood	...	...	5.7
Mixed hardwood (MH)	hardwood	...	...	0.7

Note: Ellipses indicate that data are not applicable.

vegetation map for the study region using Landsat-TM imagery. The first year that cloud-free imagery was available for the whole study area was 1987.

In choosing class categories prior to classification, rather than use all of the many vegetation and cover types in existing CALVEG data, we generalized types into broad categories that should affect fire severity. Cover types were grouped along life-form categories, with the exception of conifer and mixed conifer/hardwood, which were grouped together (Table 1). Conifer cover types were additionally grouped into broader categories (than CALVEG) of percent cover and diameter size classes. We separated forests into categories of small and medium/large trees because fire severity is largely dependent upon tree size, with larger trees normally suffering less severe effects from fire (Agee 1993). Details on the methods used to develop the 1987 vegetation map and classification accuracy results are provided in the Appendix.

CALVEG maps have generally been updated on a five-year cycle. We chose to use only the 1994 and 2004 versions, in addition to our 1987 map, since only 2% of the area that burned more than once burned between those dates. Any finer resolution would not have decreased that percentage. Vegetation and cover types in the 1994 and 2004 versions of CALVEG were re-coded to match the more general categories used for our 1987 classification (Table 1), resulting in a consistent series of three vegetation maps that could be used for analyzing trends in severity by prefire forest vegetation type, amount of tree cover, and tree diameter size class.

Fire occurrence and fire area

For analysis of fire occurrence due to ignition source regardless of fire size, we used records from FIRESTAT (*available online*),[8] which archives fire reports filed by U.S. Forest Service personnel. FIRESTAT spatial data are point features and cannot be used to carry out spatial analyses of burned area, although each report does include an estimate of fire size. FIRESTAT includes geographic locations of all ignitions regardless of size, but in our study area, it is complete only for fires since 1970 (total $N = 18\,391$ fire occurrences).

For spatial analysis on burned area and for large fires over the 1910–2008 period, we used the interagency California digital fire history database (CDF 2009), which is the most comprehensive, long-term archive of fire perimeters in the western United States. For our study area, it is considered complete for fires > 40 ha back to about 1910. Except for fires smaller than 40 ha, there is no systematic exclusion of fires from the database that would bias an analysis of trends. Fire maps of older fires were acquired and digitized to create the original database in the early 1990s (McKelvey and Busse 1996). Some perimeters for fires between 40 and 400 ha in size that occurred before 1950 are generalized, but retain size and location from fire records. Additionally, U.S. Forest Service and Department of Interior staff have spent considerable time validating and updating the database over the last decade. In our analysis, we included all fires > 40 ha that were recorded within the study area 1910–2008 (total $N = 947$) because (1) smaller fires (<40 ha) tend to be underreported, and (2) fires > 40 ha tend to represent those that escape initial attempts at containment. Fire perimeters were clipped to national forest boundaries for all analyses.

[8] http://www.fs.fed.us/fire/planning/nist/firestat.htm

Trend analyses

Time-series regression was used to examine trends over time. Common linear regression analysis of time-series data sets is often inappropriate for trend analysis of ecological variables since errors about the regression line can be autocorrelated (Edwards and Coull 1987). We therefore used Autoregressive Integrated Moving Average (ARIMA) techniques that have been used in previous studies of trends in fire effects (e.g., Stephens 2005, Miller et al. 2009*b*). We fit time domain regressions using Box-Jenkins techniques for model identification and estimation (Shumway 1988). All time-series models included a linear component and some included quadratic terms. Autoregressive terms were added when chi-square test statistics for the residuals series indicated that the residuals contained additional information that might be reasonably incorporated into a more complex model. For all time-series analyses, percentage data were square-root transformed, and number of lightning-ignited fires > 40 ha, mean and maximum fire size, and area variables were log transformed to meet normality assumptions.

Percentage of high-severity fire.—We examined the time-series trend in the percentage of fire area burning at high severity per year for the period 1987–2008, stratified by cover and vegetation type. For each year, the percentage of high-severity fire was determined by summing the area burned at high severity across all fires and dividing by the total burned area for that year. Severity data were square-root transformed, and area data were log transformed to meet normality assumptions. Due to high interannual variability in the data sets, we also graphically portray trends using an 11-year moving average of the annual data.

The percentage of high-severity fire at which a given vegetation type will burn can differ between fire events. We therefore used an Analysis of Variance (ANOVA) to examine the percentage of high-severity fire for each vegetation and cover type by fire. For areas that burned twice during the 1910–2008 period (with the second-time burned occurring during the 1987–2008 period, for which we have fire severity data), we computed the percentage of high-severity per fire stratified by cover and vegetation type for different intervals between the first and second-time burned (1–30, 31–60, and 61–98 years). Shorter intervals yielded too few hectares for some vegetation and cover types to be meaningful. A 30-year interval also closely matches what is thought to be the historic fire rotation in northwestern California (Wills and Stuart 1994, Taylor and Skinner 1998, 2003, Stephens et al. 2007). We examined the effect of previous fire on the subsequent percentage of high-severity fire in two ways. First, stratifying by vegetation and cover type (Table 1), we compared the percentage of high-severity fire between the first-time areas burned and the second-time areas burned in each of three time intervals. Second, we stratified the three major conifer vegetation types (Douglas-fir, mixed conifer, and fir/high-elevation conifers) by cover and diameter size class category (cover type categories in Table 1) and examined differences in high-severity fire for first-time burned vs. second-time burned within 30 years. We chose to examine only the first 30-year interval because the number of hectares in the remaining two 30-year intervals was too few to be stratified. Time interval, vegetation type, and cover type were considered fixed effects, while fires were a random effect in our ANOVA. Severity data were square-root transformed to satisfy normality assumptions. To satisfy the ANOVA requirement for equal variances, we applied area burned as a weight. We used post hoc tests to compare differences in means between the first-time vegetation and cover types burned, and between first-time and second-time burned. Statistical textbooks recommend that P values should be adjusted to avoid Type I errors when making multiple comparisons. However, there is significant debate in the literature whether those adjustments should be made when the data being examined are not random numbers but actual observations of ecological processes (Rothman 1990, Moran 2003, Meyn et al. 2010). We therefore base our results on non-adjusted P values, but we also report significance of P values using a Tukey-Kramer adjustment for comparison (Kramer 1956).

Patch size.—Polygons of small and medium/large conifer cover types (Table 1) were merged to form contiguous patches and patch size was limited to >900 m^2 due to the 30×30 m pixel size of Landsat imagery. Sizes of all conifer high-severity patches, and maximum conifer high-severity patch size per fire were averaged per year (giving the mean and mean and maximum patch size) and analyzed for temporal trends between 1987 and 2008 using ARIMA time-series regression.

Fire occurrence and area burned.—We used the FIRESTAT data to carry out ARIMA time-series regression for the number of all ignitions, lightning ignitions, and human-caused ignitions, for all fires and for fires > 40 ha in size. We also carried out ARIMA time-series regressions for fire occurrence and area burned from the fire perimeter data for the period 1970–2008, matching the FIRESTAT period of record. Values from both data sources were log transformed when required to meet normality assumptions.

Fire rotation.—"Fire rotation" is defined as the length of time necessary to burn an area equal to the area of interest (in this case, our study area) and is calculated by dividing the time period of interest by the proportion of the study area burned in that time period (Heinselman 1973). In spatial analyses, fire rotation is a better descriptor of fire frequency than is fire return interval for a point (Agee 1993). We calculated fire rotation for forested areas in our study region for the period 1910–2008 using a 25-year moving window in order to allow comparison of the trend in landscape-level fire frequencies over the last century with published estimates from the pre-Euroamerican settlement period. Since we did not have prefire vegetation maps for fires before 1987,

TABLE 2. Description of the abbreviations for independent variables included in the multiple regression and *t* test analyses.

Acronym	Definition	Source
DJFppt, MAMppt, JJAppt, SONppt	total precipitation 1910–2008: winter (previous Dec, Jan–Feb), spring (Mar–May), summer (Jun–Aug), fall (Sep–Nov) (summary for California (CA) north coast climate division)	WRCC (2009)
ANNppt	sum of DJFppt, MAMppt, JJAppt, and SONppt (precipitation year runs Dec–Nov)	WRCC (2009)
ANNppt1	annual precipitation lagged one year	WRCC (2009)
DJFMAMppt	winter and spring precipitation per year	WRCC (2009)
DJFmaxT, MAMmaxT, JJAmaxT, SONmaxT	mean maximum temperature 1910–2008: winter (previous Dec, Jan–Feb), spring (Mar–May), summer (Jun–Aug), fall (Sep–Nov) (summary for CA north coast climate division)	WRCC (2009)
DJFminT, MAMminT, JJAminT, SONminT	mean minimum temperature 1910–2008: winter (previous Dec, Jan–Feb), spring (Mar–May), summer (Jun–Aug), fall (Sep–Nov) (summary for CA north coast climate division)	WRCC (2009)
PDSI	Palmer Drought Severity Index 1910–2008: indexed by year and month of fire start date	NOAA (2009*a*)
$PDSI_{jja}$	mean Palmer Drought Severity Index summer (Jun–Aug)	NOAA (2009*a*)
PNA	Pacific/North American circulation pattern index per year 1950–2008: average of Jan, Feb, and previous-year Dec	NOAA (2009*b*)
IgnitionDay	Julian date of fire start	FIRESTAT
FireDuration	containment date − ignition date	FIRESTAT
NFires40	number of fires > 40 ha per year	fire perimeters
FireSize	fire size	fire perimeters
TotalBurnedYr	total area burned per year in fires > 40 ha	fire perimeters
NLightning	number of lightning ignitions per year	FIRESTAT
NIgnitions	number of all ignitions per year	FIRESTAT
CoastDist	distance to California coast from fire perimeter centroid	fire perimeters
PrismANNPrecip	annual mean precipitation at perimeter centroid	Daly et al. (1994)

we used the percentage of forested area within fires 1987–2008 as an estimate of forested area within fires prior to 1987. For this analysis, we defined forested areas as all areas classified as either conifer or hardwood vegetation types.

Underlying relationships

We examined relationships between independent variables (Table 2) and fire response variables across three different time spans:1987–2008 (fire severity data), 1970–2008 (FIRESTAT data and fire perimeter data corresponding to the FIRESTAT time period), and 1910–2008 (fire perimeter data). Percentage data were square-root transformed, and area variables were log transformed to meet normality assumptions. Data collinearity was also assessed using variance inflation factors.

Stepwise multiple linear regressions ($P_{enter} < 0.15$, $P_{remove} > 0.05$) were used to examine relationships between independent variables (Table 2), and the following variables derived from the severity data: percentage and area of fire area burned at high severity per fire (regardless of vegetation type), percentage of high-severity fire in conifer vegetation (medium/large and small diameter classes) per fire, and mean and maximum high-severity conifer patch size per fire for the 1987–2008 period ($N = 132$ fires). Any fires where we had no information on containment date or had no high-severity effects in a conifer vegetation type were dropped from the analyses. To determine whether fire-climate relationships have changed 1910–2008, we examined the relationship of the number of fires, mean and maximum fire size, and total area burned per year to seasonal and annual climate variables, the Palmer Drought Severity Index (PDSI; Alley 1984), and the Pacific/North American circulation pattern index (PNA; Wallace and Gutzler 1981) (Table 2). We divided the fire perimeter data set into three temporal groups and ran regressions for each period: early (1910–1959), late (1960–2008), and very late (1987–2008). The very late time period was selected to coincide with the fire severity data record, and the early and late periods were derived by dividing the whole period in half.

Finally, we compared differences between lighting- and human-ignition sources using two-sample *t* tests. Independent variables consisted of fire statistics from the FIRESTAT data 1970–2008 > 40 ha (lightning $N = 191$, human $N = 126$), and severity variables derived from the 1987–2008 fires (lightning $N = 102$, human $N = 28$). We used fire perimeters matching the FIRESTAT data for comparing distance between fire boundaries and the nearest Wildland–Urban Interface (WUI) or national forest boundary (USDA 2006).

RESULTS

High-severity differences among forest types

Percentage of high severity in areas that burned the first time after 1986 differed between the three major conifer types (9%, 16%, and 19% for Douglas-fir, mixed conifer, and fir/high-elevation conifers, respectively; Table 3), but there were no differences between types when re-burned within 1–30 years (10%, 12%, and 11%;

significance not shown). Among the hardwood types, deciduous oak experienced significantly higher severity than mixed hardwood the first-time burned (13% vs.8%), but was not significantly different from live oak (14%). Stratified by cover type, the percentage of high-severity fire was lower in conifer forests with trees of medium to large diameter (both closed and open classes CCM and COM) than in forests of small-diameter tress (classes CCS and COS). Among forest cover types, percentage of high-severity fire was highest in sapling/poles (SAPOL) at 16%. Percentage of high-severity fire in hardwoods (HDW) was significantly less than SAPOL, but high severity may be underreported for sprouting hardwood types since the severity data were derived from imagery acquired the year after fire. Postfire sprouting by hardwood species may reduce the apparent severity level by decreasing the difference in live biomass between post- and prefire images.

In areas that burned twice during 1910–2008, there were differences between forest and cover types when comparing the percentage of high-severity first-time burned and second-time burned (Table 3). Douglas-fir (DF) forests that had not experienced fire since at least 1910 (the beginning of our fire occurrence data set), but then burned after 1986 (i.e., during the period for which we have severity data, 1987–2008), did so at an average of 9% high-severity fire. In areas where we had record of a previous fire in DF before 1987, a second fire occurring between 1987 and 2008 tended to burn at similar levels of high severity (10%) if the fire occurred within 30 years of the first fire. Second burns occurring in DF >30 years after the initial fire were, however, significantly less severe (5% for both 31–60 years and >60 years). In contrast, the percentage of high-severity, second-time mixed conifer (MC), fir/high-elevation conifers (FIR), and closed conifer cover types (CCM and CCS) burned was generally less the than first-time burned, regardless of interval between first and second-time burned. The exceptions were MC and CCS at intervals of 61–98 years between fires (MC was close to significant at $P = 0.053$), and FIR at the 31–60 year interval (due to the small amount of area burned). Although there was generally a significant difference between first- and second-time burned, there were no differences between any of the second-time burned intervals for MC or FIR (significance not shown). For CCM the 61–98 year interval was less than the 1–30 year interval (7% vs. 10%, respectively; significance not shown). Percentage of high-severity first-time burned for any hardwood type was not significantly different than any interval the second-time burned, except for the mixed hardwood (MH) 31–60 year interval, which is likely not reliable due to the small area.

There were differences between the three major forest vegetation types when stratified by cover type (Table 4). For closed forests (CCM and CCS), DF showed no significant difference between first-time burned and second-time burned within 1–30 years. However, MC and FIR closed forests burned with a significantly lower percentage of high severity the second-time burned (within 1–30 years) than the first-time burned. Results were less clear for open forests, with only two of the six comparisons (MC small and FIR medium/large) having a significantly lower percentage of high severity when burned a second time, compared to the first (Table 4).

Temporal trends

Percentage of high-severity fire, 1987–2008.—Regardless of cover or vegetation type stratification, there were no clear trends in the percentage of high-severity fire or mean maximum high-severity conifer patch size over time. Except for mean conifer patch size, time-series regression modeling was unable to produce a model significant at $P = 0.05$ without inclusion of a quadratic component. After adding a quadratic component, three models were significant: CCS, "all forest," and "all conifer" (Table 5). The model for "all forest" is typical of models with quadratic components, where the modeled trend is unimodal and convex (Fig. 2). The interannual mean percentage of fire area experiencing high-severity fire was 25% (median = 20%); summed across the 22-year study period, 16% of the total mapped burned forested area burned at high severity. Mean percentage of high-severity fire in forested areas was inversely related to total forested area burned per year ($r^2 = 0.240$, $P = 0.039$), and the two years with the most area burned were at the beginning and end of the period (1987 and 2008). As a result, time-series models that were statistically significant included quadratic components and the "linear" trend reflected a unimodal signal, with low values of severity at each end. Mean maximum high-severity conifer patch size was positively correlated to total area burned per year ($r^2 = 0.226$, $P = 0.046$; regression not shown) and therefore had the same issue. Linear regression of mean patch size however, exhibited a decreasing trend over the period ($r^2 = 0.233$, $P = 0.042$; regression not shown).

Number of fires, mean and maximum fire size, and total area burned.—For the period 1910–2008, running 11-year averages of mean and maximum fire size and total area burned per year all reached their highest levels after 2000, but the number of fires was still below its high during the 1920s (Fig. 3). ARIMA analysis found significant autoregressive terms only for all human ignitions during the 1970–2008 period. We therefore report only linear regression results. Analysis for fires since 1970 shows that the number of fires > 40 ha, mean and maximum fire size, and total area burned per year all increased significantly ($r^2 = 0.107$ to 0.210, all $P < 0.05$). Although the number of lightning ignitions resulting in fires > 40 ha increased ($r^2 = 0.185$, $P = 0.006$), the total number of lightning ignitions remained constant. Conversely, the total number of human ignitions fell ($r^2 = 0.265$, $P < 0.001$), while the number of human ignitions resulting in fires > 40 ha remained constant.

TABLE 3. Total number of hectares burned and ANOVA estimates with SE for percentage of high severity, by fire, for forest vegetation and cover types for all fires >400 ha for the 1987–2008 period.

Type	All fires			All first-time burned†			All first-time burned, significantly different‡
		High severity			High severity		
	Area (ha)	%	SE	Area (ha)	%	SE	
Forest vegetation type							
Douglas-fir (DF)	207 910	9	0.15	154 372	9	0.15	GP, *MC, FIR, DO, LO*
Gray pine (GP)	1 550	14	0.44	1 243	17	0.49	DF, MH
Mixed conifer (MC)	171 939	15	0.16	139 878	16	0.16	*DF, FIR, LO*, MH
Fir/high-elevation conifer (FIR)	55 478	18	0.15	46 992	19	0.15	*DF, MC, DO, LO, MH*
Deciduous oak (DO)	27 730	14	0.17	24 434	13	0.18	*DF, FIR*, MH
Live oak (LO)	37 153	14	0.17	26 572	14	0.17	*DF, MC, FIR*
Mixed hardwood (MH)	4 650	11	0.28	2 727	8	0.35	GP, MC, *FIR*, DO
Forest cover type							
Conifer closed medium/large (CCM)	258 308	12	0.15	202 349	13	0.15	CCS, COS, *SAPOL*
Conifer closed small (CCS)	85 448	14	0.15	61 941	15	0.16	CCM, COM
Conifer open medium/large (COM)	41 008	12	0.16	33 994	13	0.17	CCS, COS, SAPOL
Conifer open small (COS)	52 113	14	0.16	44 205	14	0.16	CCM, COM
Hardwood (HDW)	69 533	14	0.15	53 735	14	0.16	SAPOL
Sapling/poles (SAPOL)	74 483	16	0.15	50 645	16	0.16	*CCM*, COM, HDW

Notes: For areas that burned twice during the 1910–2008 period and the second time within the 1987–2008 period for which fire severity data are available, the data are broken down into different intervals between the first and the second burn. Less than 4% of the study area burned three or more times during the 1910–2008 period; data for more than two burns are therefore not shown. Fires were assumed to be random effects, while time and vegetation and cover types were fixed effects in the ANOVA. Displayed percentage high-severity data are squared since the data were square-root transformed before running the ANOVA.

† First time an area burned after 1910; does not include areas that burned before 1987.

‡ Differences of least-squares means, $\alpha = 0.05$, $P < 0.05$; when italicized, the P values were significant using the Tukey-Kramer adjustment.

§ Boldface indicates significantly different from "All first-time burned"; differences of least-squares means, $\alpha = 0.05$, $P < 0.05$; when italicized, P values are significant using the Tukey-Kramer adjustment.

¶ Estimate not significant at $P < 0.05$.

Fire rotation.—Approximately one-third of the study area burned between 1910 and 2008. Excluding water and barren areas, 33% burned at least once, 7% at least twice, and 1% three or more times. Fire rotation in forested areas in 1934 (the end of the first 25-year interval) was 267 years and rotation period steadily increased to a peak value of 974 years in 1984 (Fig. 4). Rotation values fell to 256 years in 1987, the year with the second most area burned (Fig. 3d), and continued to fall to 95 years in 2008, the year with the most area burned. The rotation period for high-severity fire is somewhat more problematic because high-severity fire tends to occur more often in some locations than others due to topographic variation (e.g., elevation and aspect; Taylor and Skinner 1998), and therefore, the expected rotation would be dependent upon geographic location. However, simply calculating high-severity rotation by dividing rotation period for all fire in forested areas by percentage high-severity fire measured 1987–2008 as a first-order estimate (95/0.16), current rotation for high-severity fire over the last 25-year period for the entire study region (i.e., ignoring important geographic variability) would be somewhere around 600 years. We urge caution in interpretation of this number. Obviously, the temporal trend in high-severity rotation would follow the same decreasing trajectory as the overall fire rotation.

Underlying relationships

Percentage of high-severity and fire size.—In conifer vegetation types within individual fires mean and maximum high-severity patch size, and percentage of high-severity fire tended to be greater: (1) with larger fire size, (2) in fires that ignited later in the year, and (3) in years when less area burned across the study region (Table 6). The percentage of high-severity per fire was inversely related to spring precipitation. Larger patches of high-severity fire tended to occur in forests closer to the Pacific coast. Although the percentage of high-severity per fire was more strongly (negatively) related to total area burned per year than fire size, the area of high-severity and maximum high-severity patch size in each fire were more strongly (positively) related to fire size.

Climate 1910–2008.—Mean annual precipitation increased by 3.00 mm/yr for a total of 30 cm between 1910 and 2008 ($r^2 = 0.069$, $P = 0.009$; regression not shown); however, when precipitation was broken down by seasons, none of the seasonal trends were statistically significant (but for spring precipitation; $P = 0.065$). Winter maximum and all seasonal minimum temperatures increased over the time period, led by summer minima (+1.63°C, $r^2 = 0.426$, $P < 0.001$; regression not shown). Splitting the data set into early (1910–1959) and later (1960–2008) periods, correlations of fire variables (number of fires, fire size, maximum fire size, and area

Table 3. Extended.

Second-time burned 1910–2008, by interval since first burn§								
1–30 yr			31–60 yr			61–98 yr		
	High severity			High severity			High severity	
Area (ha)	%	SE	Area (ha)	%	SE	Area (ha)	%	SE
22 158	10	0.19	7787	***5***	0.24	9914	***5***	0.22
2	**1**¶	1.95	97	8	1.37	181	8	1.23
9724	**12**	0.31	5505	**12**	1.30	7272	13	0.38
3524	**11**	0.22	160	10	0.27	2155	**9**	0.24
850	12	0.59	1178	12	0.50	782	14	0.61
2613	8	0.35	3047	11	0.33	2064	9	0.39
291	12	0.97	440	***27***	0.80	766	5	0.61
19 170	**10**	0.18	7084	***8***	0.23	11 965	*7*	0.20
9453	***10***	0.22	4322	*7*	0.28	4763	12	0.27
3541	14	0.30	758	**6**	0.58	1257	10	0.46
3243	13	0.31	1386	9	0.44	1537	14	0.42
3755	12	0.29	4665	13	0.27	3612	11	0.30
10 413	13	0.21	4911	**11**	0.27	3226	16	0.31

burned) shifted from inverse correlations with PDSI in the early period, to inverse correlations with summer precipitation in the later period (Table 7). Summer precipitation dipped (slightly) below average from 1999–2008, the same period during which mean and maximum fire size, and total area burned were at their highest levels. Mean summer precipitation was lowest during the 1920s, yet the number of fires was the only fire statistic that was larger during this period than during 1999–2008. Even though summer precipitation was low during the 1999–2008 period, mean annual precipitation during this period was more or less equal to the 99-year average (mean annual precipitation 1999–2008 divided by 99-year mean = 0.992). The strength of the association between fire variables and climatic parameters increased markedly from the beginning of the study period to the end (Table 7). Variance in fire statistics explained by the climate regressions increased from an R^2 range of 0.268 to 0.366 in the 1910–1959 period, to an R^2 range of 0.413 to 0.599 in the 1987–2008 period (adjusted R^2 values).

Lightning vs. human ignitions.—For fires $>$ 40 ha, individual fire size, duration, number of fires, and total area burned per year were all significantly higher for lightning-ignited fires than for human-ignited fires; yet fire severity was lower in lightning-caused fires (Table 8). Human-ignited fires occurred closer to WUI or national forest boundaries than lightning-ignited fires. Between human- and lightning-caused fires, there were no significant differences in mean (8 vs. 11 ha, respectively) or maximum (106 vs. 97 ha, respectively) high-severity patch size or ignition dates. Lightning fires tended to occur under drier climatic conditions than did human-caused fires, even though both tended to occur under drier than average conditions (mean PDSI = −1.83 and –0.78, and PNA = 0.49 and 0.33 for lightning and human ignitions, respectively).

Discussion

In northwest California we found no clear temporal trend in the percentage of forest area burning at high severity for the period 1987–2008. However, during the same period, mean and maximum fire size, and total annual area burned all increased to levels above any recorded since the U.S. Forest Service began keeping records at the beginning of the 20th century. Despite this increase in fire size and area burned, the annual number of fires remains below the maximum in the 1920s.

Fire occurrence and fire area (1910–2008)

Our results suggest that broadly similar trends in fire number, size, and annual area burned are occurring across the mountains of central and northern California. In the Sierra Nevada, the eastern and southern California Cascades and our study region (which includes the Klamath Mountains, north Coast Ranges, and northern and western California Cascades), fire size and annual burned area now respond most directly to precipitation during the fire season, and fire number, size, and annual burned area have been rising even as regional precipitation increases (Miller et al. 2009*b*). Our results also suggest that the nature of the fire–climate relationship in northwest California has changed since the beginning of the 20th century. In our study, the number of fires, fire size, and area burned were most strongly (negatively) associated with $PDSI_{jja}$ early in the study period, but became most strongly (negatively) associated with summer precipitation at the end of the period. $PDSI_{jja}$ integrates temperature and precipitation anomalies over the summer months (but it has a lag built in so that it does not respond to short-term inputs of small amounts of precipitation [Herweijer et al. 2007]). Thus, similar to the Sierra Nevada (Miller et al.

TABLE 4. Total number of hectares burned and ANOVA estimates, with SE, for percentage of high severity, by fire, for forest vegetation type stratified by cover type for all fires >400 ha for the 1987–2008 period.

Forest vegetation type and cover type	All first-time burned†			All first-time burned, significantly different‡	Second-time burned within 1–30 yr§		
		High severity				High severity	
	Area (ha)	%	SE		Area (ha)	%	SE
Douglas-fir (DF)							
Conifer closed medium/large (CCM)	99 921	8	0.16	DFccs, *MCccm, MCccs, MCcom, MCcos, FIRccm, FIRccs, FIRcom, FIRcos*	12 778	8	0.20
Conifer closed small (CCS)	33 698	10	0.17	DFccm, DFcom, *MCccm, MCccs, MCcom, MCcos, FIRccm, FIRccs, FIRcom, FIRcos*	7511	8	0.22
Conifer open medium/large (COM)	8901	8	0.21	DFccs, *MCccm, MCccs, MCcom, MCcos, FIRccm, FIRccs, FIRcom, FIRcos*	890	13	0.46
Conifer open small (COS)	11 852	9	0.19	*MCccm, MCccs, MCcom, MCcos, FIRccm, FIRccs, FIRcom, FIRcos*	980	11	0.44
Mixed conifer (MC)							
Conifer closed medium/large (CCM)	75 812	15	0.16	*DFccm, DFccs, DFcom, DFcos*, MCccs, MCcos, *FIRccm, FIRccs*	4895	**10**	0.24
Conifer closed small (CCS)	23 825	17	0.17	*DFccm, DFccs, DFcom, DFcos*, MCccm, MCcom, *FIRccm, FIRccs*	1577	**11**	0.36
Conifer open medium/large (COM)	18 456	14	0.18	*DFccm, DFccs, DFcom, DFcos*, MCccs, MCcos, *FIRccm, FIRccs*, FIRcom	1680	15	0.35
Conifer open small (COS)	21 785	17	0.18	*DFccm, DFccs, DFcom, DFcos*, MCccm, MCcom, FIRccs	1572	**12**	0.36
Fir/high-elevation conifer (FIR)							
Conifer closed medium/large (CCM)	26 552	19	0.17	*DFccm, DFccs, DFcom, DFcos, MCccm*, MCccs, *MCcom*, FIRcos	1497	***6***	0.39
Conifer closed small (CCS)	4275	23	0.25	*DFccm, DFccs, DFcom, DFcos, MCccm*, MCccs, *MCcom*, MCcos, FIRcom, FIRcos	365	**6**	0.70
Conifer open medium/large (COM)	6391	17	0.22	*DFccm, DFccs, DFcom, DFcos*, MCcom, FIRccs	971	**10**	0.44
Conifer open small (COS)	9775	16	0.20	*DFccm, DFccs, DFcom, DFcos*, FIRccm, FIRccs	691	15	0.51

Notes: For areas that burned twice during the 1910–2008 period and the second time within the 1987–2008 period for which fire severity data are available, the data are displayed when the interval between the first and the second burn was <30 years. Fires were assumed to be random effects while vegetation type stratified by cover type and time were fixed effects in the ANOVA. Displayed percentage high-severity data are squared since the data were square-root transformed before running the ANOVA.

† First time an area burned after 1910; does not include areas that burned before 1987.

‡ Differences of least-squares means, $\alpha = 0.05$, $P < 0.05$; when italicized, P values were significant using the Tukey-Kramer adjustment.

§ Boldface indicates significantly different from "All first-time burned"; differences of least-squares means, $\alpha = 0.05$, $P < 0.05$; when italicized, P values are significant using the Tukey-Kramer adjustment.

2009*b*), a temperature-related variable was most important for explaining fire statistics during the early part of the study period, while fire-season precipitation on its own was the best predictor by the end of the period. Also like the Sierra Nevada (Miller et al. 2009*b*), the strength of the fire–climate relationship in northwestern California increased markedly from the beginning of the 20th century to the beginning of the 21st century (Table 8). Our results provide yet further confirmation of the growing importance of climate in regulating wildfire across the western United States (e.g., Field et al. 1999, Brown et al. 2004, Running 2006, Westerling et al. 2006, Morgan et al. 2008, Littell et al. 2009, Miller et al. 2009*b*).

Another intriguing trend in northwestern California is a strong temporal increase in the importance of lightning fires in the region. In the early part of the 20th century, lightning accounted for ~42% of area burned in all recorded fires, but by the end of the century, 87% of area burned was caused by lightning. A similar trend has been reported for the Sierra Nevada, where area burned by lightning-caused fires changed

TABLE 5. Results for time-series regression analyses of the percentage of high-severity fires and mean high-severity patch size for the 1987–2008 period.

Model statistic	Conifer closed small (CCS)	All forest†	All conifer‡	Conifer mean patch size
N	18	18	18	18
Error degrees of freedom	12	12	12	16
Parameter estimates				
Sigma squared (model variance)	1.018	0.719	0.004	0.436
Intercept	−30.229	−23.029	−1.933	3.935
Linear	−1.790	−1.422	−0.115	−0.052
Quadratic	−0.022	−0.018	−0.001	...
Autoregressive function (AR) 1	−0.159	−0.201	−0.254	...
AR2	−0.270	−0.296	−0.313	...
AR3	−0.674	−0.691	−0.729	...
P (linear)	0.001	0.001	0.000	0.042
P (quadratic)	0.001	0.001	0.000	...
P (AR1)	0.441	0.379	0.223	...
P (AR2)	0.133	0.107	0.078	...
P (AR3)	0.003	0.003	0.001	...
Statistics of fit				
Mean-square error (MSE)	34.630	23.959	0.140	9.094
Root mean-square error (RMSE)	0.913	0.792	0.061	0.622
Mean absolute percentage error (MAPE)	14.296	13.592	15.297	16.473
Mean absolute error (MAE)	0.691	0.650	0.047	0.511
R^2	0.567	0.529	0.528	0.233
Adjusted R^2	0.386	0.333	0.331	0.185
Akaike Information Criterion	8.715	3.588	−88.939	−13.069
Schwarz Bayesian Information Criterion (SBC)	14.057	8.930	−83.597	−11.288

Note: Ellipses indicate that data are not applicable.
† "All forest" includes hardwood and conifer cover types, excluding sapling/poles.
‡ "All conifer" includes only conifer cover types, excluding sapling/poles.

from 5% to >40% over the 20th century (Weatherspoon and Skinner 1996). We suggest the increasing importance of summer precipitation later in the study period may be related to the increasing dominance of lightning-caused fires. What little precipitation occurs in this Mediterranean climate during the summer months is largely the result of occasional convective thunderstorms that also produce the lightning that is the source of most of the area burned. Annual precipitation in northwestern California is increasing, but almost entirely during the October to May rainy season (i.e., not during the summer fire season); however, our analysis suggests that amount of precipitation at the time of ignition has become more important in recent decades than seasonal drought in driving fire activity and fire area. Ironically, dry years in which large areas burn due to lightning-

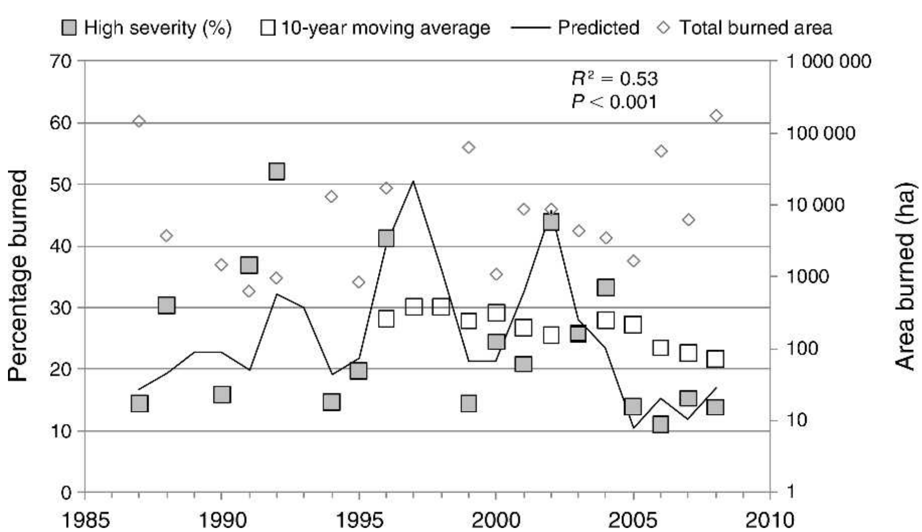

FIG. 2. Temporal trend in the percentage of area burned at high severity for all forest types (excluding sapling/poles) combined in the study region, 1987–2008, with the best-fit regression function, 10-year moving average for percentage of high-severity fire, and total forested area burned (right-hand *y*-axis, log scale). The *P* value shown refers to the linear trend. The data were best fit by a combination of linear, quadratic, and third-order autoregressive function.

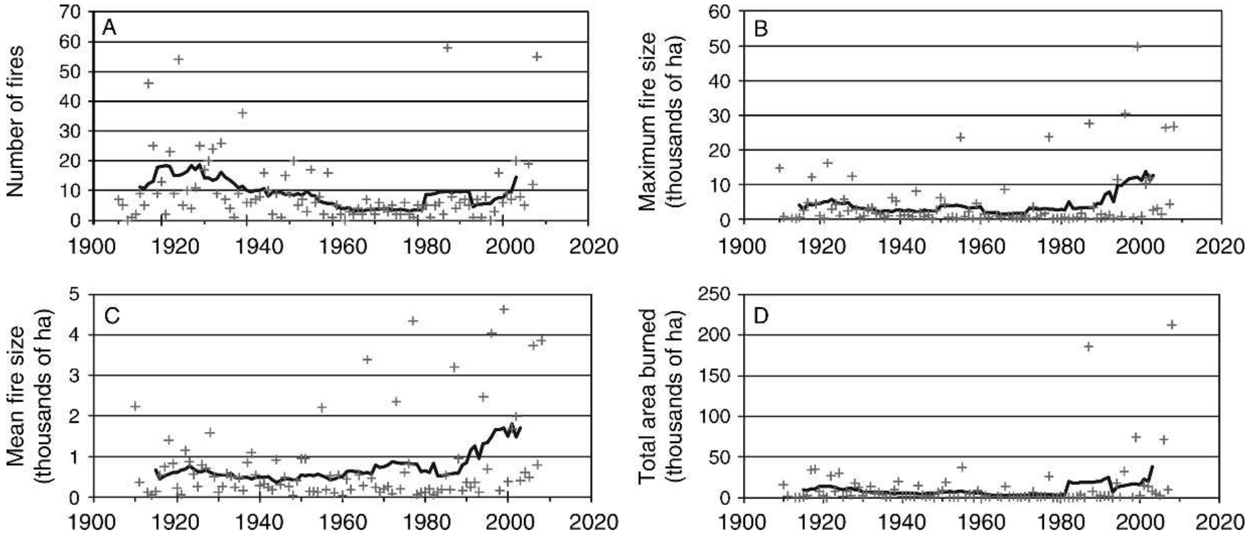

FIG. 3. (A) Number of fires, (B) maximum fire size, (C) mean fire size, and (D) annual total burned area for fires >40 ha for the 1910–2008 period within the study region (indicated by crosses). Fire sizes are for area within forest boundaries. The 11-year moving averages (solid line) are only for display purposes. The four years (1987, 1999, 2006, and 2008) with the most area burned per year were years with widespread lightning events.

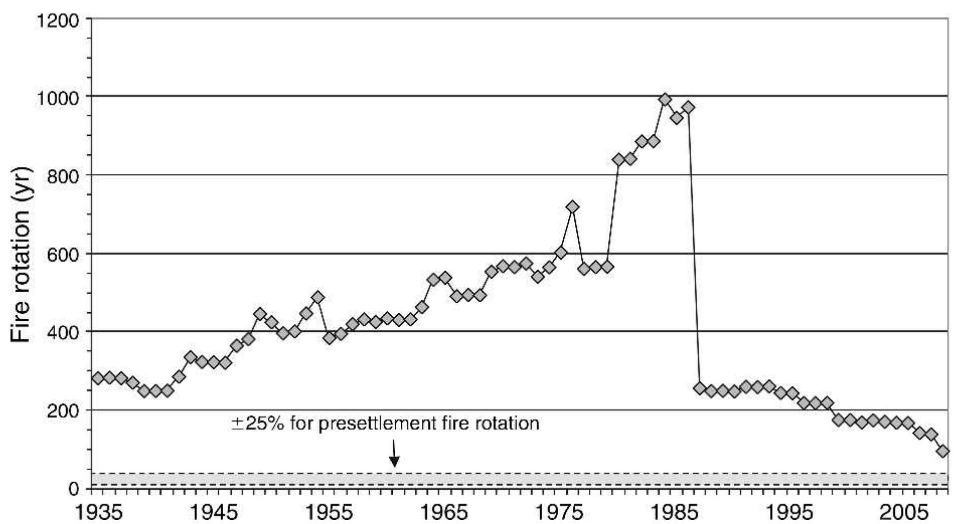

FIG. 4. Fire rotation for forested areas for the period 1910–2008. Values are calculated for the sum of forest area burned during the prior 25 years. For example, the point at year 2000 represents the fire rotation calculated for the entire period 1976–2000. The dashed lines represent the estimated presettlement fire rotation for northwestern California forests (Taylor and Skinner 1998, 2003).

TABLE 6. Multiple regression results showing the standardized partial regression coefficients for a series of independent variables potentially influencing fire effects variables for fires >400 ha.

Dependent variable	MAMmaxT	MAMppt	IgnitionDay	FireSize	TotalBurnedYr	CoastDist	P	R^2	Adjusted R^2
Fire % high severity	···	−0.212	0.238	0.231	−0.494	···	<0.001	0.252	0.227
Conifer % high severity	···	−0.222	0.242	0.238	−0.492	···	<0.001	0.254	0.230
Mean conifer patch size	···	···	0.197	0.226	−0.180	−0.189	<0.001	0.148	0.120
Maximum conifer patch size	···	···	0.171	0.658	−0.161	−0.168	<0.001	0.500	0.484
Fire high-severity area	0.110	···	···	0.873	−0.219	···	<0.001	0.722	0.716

Notes: Standardized partial regression coefficients significant at $P < 0.05$ are shown. See Table 2 for a description of independent variables. Ellipses indicate that data are not applicable.

Table 7. Multiple regression results showing the standardized partial regression coefficients for climate variables potentially influencing number of fires, fire size, and total area burned per year for fires >40 ha for the 1910–2008 period.

Dependent variable and period	DJFmaxT	SONminT	JJAppt	SONppt	$PDSI_{jja}$	P	R^2	Adjusted R^2
Number of fires								
1910–1959	…	…	…	…	−0.615	<0.001	0.378	0.365
1960–2008	…		−0.280	−0.335	−0.368	<0.001	0.369	0.336
1987–2008	…	0.310	−0.797	…	…	<0.001	0.637	0.599
Mean fire size								
1910–1959	−0.300	…	…	…	−0.513	<0.001	0.328	0.300
1960–2008	…	…	…	…	−0.394	0.005	0.155	0.137
1987–2008	…	…	−0.664	…	…	<0.001	0.441	0.413
Maximum fire size								
1910–1959	…	…	…	…	−0.532	<0.001	0.283	0.268
1960–2008	…	…	…	…	−0.416	0.003	0.173	0.155
1987–2008	…	…	−0.712	…	…	<0.001	0.507	0.483
Total annual area burned								
1910–1959	…	…	…	…	−0.616	<0.001	0.380	0.366
1960–2008	…	…	…	−0.265	−0.480	<0.001	0.288	0.258
1987–2008	…	…	−0.748		…	<0.001	0.559	0.537

Notes: Standardized partial regression coefficients significant at $P < 0.05$ are shown. See Table 2 for description of independent variables. Ellipses indicate that data are not applicable.

ignited fires are often years of relatively little lightning activity. Because summer precipitation is almost entirely convective in nature, years with greater lightning activity are also those with greater summer precipitation, which results in less area burned (Skinner et al. 2006). In northwestern California, the outcome of these relationships is that years that produce fewer lightning strikes account for most of the area burned by lightning-caused fires (Skinner et al. 2006). This is similar to results found in the Pacific Northwest (Washington and Oregon), where the number of lightning strikes was not found to be directly proportional to the number of fire starts (Rorig and Ferguson 2002).

We propose that the increasing importance of rainfall during thunderstorms to subsequent fire activity late in the study period may be partly due to its effect on the ability of fire crews to access fires and suppress them during their initiating stage (we should also acknowledge that the last half of the study period has seen dramatic improvements in the ability of fire crews to more quickly access much of this rugged region, on the ground and especially in the air). When a lightning ignition is accompanied by rain, fire spread and intensity are reduced and first response crews have more time to reach the ignition area before the fire becomes too large to control. Therefore, wetter conditions at the time of ignition lead to fewer fires that escape initial attack. Thus, the amount of precipitation accompanying summer thunderstorms can have both a direct and indirect influence on the resulting amount of area burned (Rorig and Ferguson 1999, Rorig et al. 2007, Skinner et al. 2006).

The greater importance of the PDSI early in the study period may also be due in part to the poor access that fire fighters had to wildland ignitions earlier in the 20th century. Even when rain accompanied lightning starts,

Table 8. Differences between lightning- and human-ignited fires.

Variable	Period	Mean difference (lightning − human)†	SE	P
Fire % high severity	1987–2008	−1.994	0.294	<0.001
Conifer % high severity	1987–2008	−1.916	0.318	<0.001
Mean conifer patch size (ha)	1987–2008	−0.080	0.069	0.315
Maximum conifer patch size (ha)	1987–2008	0.024	0.147	0.878
Distance to WUI or forest boundary (m)	1970–2008	1753.200	412.640	<0.001
FireSize (fires > 40 ha)	1970–2008	0.351	0.087	<0.001‡
FireDuration (fires > 40 ha)	1970–2008	27.973	3.634	<0.001‡
IgnitionDay (fires > 40 ha)	1970–2008	−0.900	5.300	0.865‡
PDSI (fires > 40 ha)	1970–2008	−1.045	0.195	<0.001‡
PNA	1970–2008	0.158	0.064	0.016
NFires40	1970–2008	0.605	0.043	<0.001
TotalBurnedYr	1970–2008	1.265	0.085	<0.001‡

Note: See Table 2 for a description of variables; WUI stands for Wildland–Urban Interface.
† Two-sample t test, H_0 $(\mu_1 - \mu_2) = 0$.
‡ F test indicated equal variances; P values are for unequal variances otherwise.

difficulties in access may have slowed crew arrival on site sufficiently that the effect of moisture from the thunderstorm was largely gone before arrival. In other words, longer suppression response times would reduce the effectiveness of the window of reduced fire behavior that would have resulted from convective rainfall accompanying the lightning start. This would allow live and dead fuel moistures (which are typically very low in the California summer) to play a greater role in subsequent fire behavior and fire suppression effectiveness.

With the shift in the association of fire statistics from PDSI to summer precipitation, the annual number of fires escaping initial attack within our study has increased during the last portion of the 20th century, but overall numbers still remain below historical levels. In concert with the increase in number of fires, total area burned is moving closer to presettlement levels. Fire rotation in forested areas decreased from around 267 years early in the 20th century to about 95 years in 2008, primarily due to the increase in total area burned during 1987–2008 (Figs. 3 and 4). Importantly, even this reduction to a 95-year fire rotation is still three to six times longer than estimates of past fire rotation reported from the literature (Taylor and Skinner 1998, 2003, Stephens et al. 2007). Although there has been a shift in fire management policy in recent years to manage some fires for ecological benefit (e.g., not fully suppressing fires that are burning under more benign weather and fuel conditions), primarily in wilderness areas, the practice of suppressing most ignitions remains. It is, therefore, evident that the increase in area burned is at least partially due to more ignitions escaping initial attack and fires growing large, even under full suppression efforts.

Fire severity (1987–2008)

The four years during 1910–2008 when the most area burned were 1987, 1999, 2006, and 2008, all of which are included in our fire severity data set. Moreover, the two years with the most area burned bracket the period of our severity data set. The mean area burned in those four years is statistically larger than the remaining 95 years (two sample *t* test, $P = 0.02$). In each of these years, fires were primarily caused by widespread lightning events that severely strained fire suppression resources. Overall fire severities were relatively low due to burning for weeks to months through variable, often moderate meteorological conditions and the fact that many of the fires burned well into the fall (see next paragraph). It is probable that the coincidental timing of our severity data set with the two years with the most area burned in a century precluded us from detecting any underlying trend in percentage of high-severity fire. For example, removing 1987 and 2008 from the time-series analysis results in a significant decreasing linear trend without a quadratic component for "all forest" ($R^2 = 0.557$, $P = 0.046$; regression not shown). Our regression results underscore how widespread lightning events were a primary influence on severity over the 22-year period of our severity data.

Our findings in northwest California differ from results for the Sierra Nevada and adjacent southern Cascades, where an increasing trend in the percentage of high-severity fire for some forest types has been reported (Miller et al. 2009*a*). Differences between regions may be explained by differences in topography of northwestern California when compared with the Sierra Nevada or the southern Cascades. While the topography in both northwestern California and the Sierra Nevada are extremely rugged, canyons in the Sierra Nevada and the relatively broad ridgetops tend to align with westerly winds, whereas the topography in northwestern California is much more dissected, leading to a formation of temperature inversions. In addition, road access tends to be more limited in northwestern California, making fire suppression efforts difficult. Consequently, years with multiple contemporaneous ignition events (e.g., 1987, 1999, 2006, and 2008) can produce swarms of large fires that burn for months, many into late fall when weather conditions aid in the final containment of the fires.

In northwest California, fire intensity historically was lowest on lower slopes and north- and east-facing aspects, and greater on mid- and upper-slope positions, especially on south- and west-facing aspects, where higher temperatures and afternoon winds promote drier conditions (Weatherspoon and Skinner 1995, Taylor and Skinner 1998, Alexander et al. 2006). Long-term temperature inversions under stable air masses that are common within the region during the summer can trap smoke in valleys, leading to cooler temperature and higher humidity, and resulting in less severe fire effects at lower slope positions (Robock 1988, 1991). Reduced fire intensity, less crowning, and more surface fire are more common under temperature inversions. When temperature inversions erode, large areas of high-severity fire can occur due to higher temperatures and increased winds (e.g., Megram fire 1999, Motion and Panther fires 2008). Very strong inversion effects characterized much of the 1987 and 2008 fire years, resulting in lower than average fire severity even though 100 000's of hectares of forest burned.

Like other studies in northwest California, we found that fire severity was lower in stands dominated by medium and large trees than in stands dominated by small trees (Weatherspoon and Skinner 1995, Jimerson and Jones 2003, Odion et al. 2004, Alexander et al. 2006). We also found that areas that had experienced fire in the recent past also influenced severity, but here our results appear to differ from some other reports. Based upon a set of fires from a single year (1987), Odion et al. (2004) reported that, when closed forests burned twice after 1920 (more recently re-burned), they burned with more than double the percentage of high severity than did closed forests that previously burned before 1920 (long unburned). However, we found that closed canopy medium/large (CCM) forests that burned a second time after 1910, burned with a lower percentage of high severity per fire than did when any CCM forest burned the first time. Our results may differ because Odion et al. (2004) did not examine differences due to tree diameter size class, nor did they account for variability between fires. We also show that severity is influenced by forest vegetation type, but Odion et al. did not distinguish between types.

When stratifying by vegetation type, we found differences in the percentage of high-severity fire between forests burned the first or the second time. For Douglas-fir, high severity was not significantly different per fire the first-time burned vs. the second-time burned within the first 30 years, but when the interval between first and second fire was >30 years, high-severity fire was significantly less than first-time burned (5% vs. 9%). In contrast, severity in mixed conifer was less when burned a second time compared with first-time burned (12–13% depending upon interval vs. 16%).

It should be noted that the percentage of high-severity fire we report for "All first-time burned" in Tables 3 and 4 were for areas that burned between 1987 and 2008, but without a previously recorded fire since the beginning of record keeping in 1910. These areas were "long unburned" by Odion et al.'s definition since they had not burned for a minimum of between 76 and 98 years, and probably longer. Therefore, the high-severity percentages for "all first burned" are more likely to be a better representation of "long-unburned" forests than are the percentages for the 61–98 year interval due to the much larger area of "all first burned." Importantly though, "long unburned" forests are probably not characteristic of presettlement conditions for the majority of Douglas-fir and mixed-conifer forests in our study area, since historical mean fire return intervals are thought to have been <30 years (Wills and Stuart 1994, Taylor and Skinner 1998, 2003, Skinner et al. 2006).

Fire behavior is influenced by amount and arrangement of fuels that are often associated with vegetation type (Agee 1993, Sugihara et al. 2006). For example, short fir needles compact more densely than do longer pine needles, which influences surface fuel combustibility (Pyne et al. 1996). Where fire has been excluded for many years, less severe effects are frequently noted in mature Douglas-fir forests than in mature ponderosa pine forests, presumably because of differences in the fuel bed and the thick bark of mature Douglas-fir (Weatherspoon and Skinner 1995, Skinner et al. 2006). Conversely, protective bark develops earlier in the life of ponderosa pine than in Douglas-fir, leaving young Douglas-fir more susceptible to fire damage than equivalent-aged ponderosa pine (Skinner et al. 2006). Therefore, any examination of severity patterns based solely upon forest density and size class can result in erroneous conclusions.

We believe the severity pattern we see for first- and second-time burned associated with CCM Douglas-fir is likely characteristic of areas with increased fuels due to fire suppression. Our data show that where areas that were long unburned but recently burned twice, the proportion of high-severity fire was not significantly different the second-time burned within 1–30 years (Table 4). We hypothesize that small diameter trees and understory shrubs that are killed but not consumed in the first low-severity fire become dried, and if a second fire enters the same area before the dead fuels decompose, these fuels could contribute to higher intensities than would otherwise occur. Severity in the second burn is influenced by the time since the stand last burned and the rate that dead fuels from the first fire decompose, leading to reduction of fuel load and an increase in fuel bed bulk density as fuels decompose and compact over time. Thus, there is a corresponding decrease in the proportion of high-severity fire when the interval between fires increases (Table 3; Yin 1999). Due to the limited area in our data set that burned three times, we could not explore how the proportion of high-severity would change upon a third entry by fire.

Two factors should be considered when interpreting the re-burn analysis. We do not know the severity in fires prior to 1987, nor did we track the history of individual sites and therefore did not determine how severity in prior fires altered forest structure, thereby affecting severity in subsequent fires. We have confidence that areas mapped as CCM in our prefire maps were forested for the entire time period. These areas were most likely not altered by severe fire or they would not have retained the CCM state in the prefire maps. The degree and time frame that other areas were altered is unknown. For example, areas classed as sapling/poles (SAPOL) obviously did not retain that type throughout the full 1910–2008 period. Instead, they represent areas that either burned at high severity the first time they burned (possibly prior to 1987), and then were planted or re-established from seed before burning a second time, or were harvested and planted prior to burning.

Thompson and Spies (2009) found that areas with open tree canopies and high levels of understory shrub cover were associated with the highest levels of tree crown damage in the 2002 Biscuit fire, which also partially occurred in our study area. Our vegetation data do not provide the amount of shrub cover in forested areas, but our results indicate severity for conifer forests the first-time burned was not higher in open than closed canopy areas of the same diameter class (Tables 3 and 4).

One complication with interpreting our severity results for open stands is that our severity data are primarily an indication of change in chlorophyll. As a result, the high-severity rating for open forests may be more indicative of a change in broad-leafed shrub understory that would have a stronger chlorophyll signature than the overstory conifers. In very open forests we may not detect whether overstory conifers survived, and therefore, the severity rating may only apply to the understory shrubs, not the conifers. Additionally, our severity ratings are derived from one-year postfire imagery, and any resprouting shrubs would tend to lower the severity rating in comparison to an immediate postfire assessment.

In the fires we analyzed, the percentage of fire area that experienced high severity was greater for human-ignited fires than for lightning-ignited fires. There are multiple indications that this relationship may be due to more aggressive suppression of human-ignited fires. First, although it is possible that human-ignited fires that escaped initial attack and became large enough to be counted in this data set are those that occurred under weather conditions more likely to lead to high-severity effects (and after weather conditions moderate the fires are contained), there was no evidence that human-ignited fires occurred under seasonal climate conditions that would lead to more intense fires on average than lightning fires. There was no difference between average ignition date, and more human-caused fires actually tended to occur in fire seasons that were less dry. Second, most of the large lightning fires occurred during years when there were many simultaneous ignitions across a broad area and total burned area was higher. Under these conditions (widespread, simultaneous ignitions of multiple lightning fires), fewer fire-fighting resources are available to suppress each fire than in the average human-ignition scenario, in which fires tend to occur in (relative) temporal isolation (Haight and Fried 2007). Third, the percentage of high-severity fire was also inversely related to total area burned per year. To understand the potential bias due to fire years with the most area burned, we eliminated five years with at least 10 large fires resulting from lightning ignitions (1987, 1999, 2003, 2006, and 2008; accounting for 63% of the total area burned) and reran the *t* test analysis. The percentage of high severity per fire was still significantly different between lightning-and human-ignited fires ($P = 0.003$). Fourth, fires that occur in close proximity to access corridors and urban areas tend to be of high concern, and therefore aggressively suppressed, and centroids of human-ignited fires were significantly closer to the WUI or forest boundary on average (lightning = 3.6 km vs. human = 1.9 km; Table 8). Finally, although lightning-ignited fires were generally larger than human-ignited fires, there was no difference in either mean or maximum conifer high-severity patch size between human- and lightning-ignited fires.

Due to the strong effects of topography, site conditions and other factors on fire severity in northwestern California, we hesitate to estimate a region-wide fire rotation for high-severity fire, but a simple ballpark mean would be around 600 years. This is much lower than the 1740 year high-severity fire rotation calculated for the Klamath Mountains by Odion et al. 2010. There are several reasons for our lower rotation period. First, Odion et al. (2004) and Odion et al. (2010) are based on a more limited data set, both in terms of spatial extent and time. Their data cover only a portion of the Klamath region and are derived from a few fires in a single extraordinary fire year of extensive lightning-caused fires (1987). Average fire severity in 1987 was generally low due to a number of factors, including the late season of burning and a long period of temperature inversions typical of fall conditions. Second, Odion et al. 2010 define "forest" to be composed only of closed-canopy stands of medium and large trees. In contrast, our definition of forest follows U.S. Forest Service national classification standards that include both small trees and areas with at least 10% tree cover (Brohman and Bryant 2005). Third, we calculated rotation through 2008 (vs. 2007 used by Odion et al. 2010), which gave us a shorter rotation period, since 2008 saw the most area burned in a single year within the study period (Fig. 3d). Finally, we use a 25-year vs. Odion et al.'s 58-year period to calculate rotation. We believe our calculation is more representative of the current fire rotation due to the increasing trend in fire area burned during the last 100 years (Fig. 3).

Conclusions

Our study used the most comprehensive information available to date to investigate contemporary temporal and spatial fire patterns in northwestern California. Our data sets encompassed the entire data record for fire perimeters for medium and large fires, the entire data record for fire occurrence, and the entire data record for standardized, remotely sensed fire severity. Other studies from the region have based their conclusions on analyzing fires from one or a few years (e.g., Weatherspoon and Skinner 1995, Jimerson and Jones 2003, Odion et al. 2004, 2010, Alexander et al. 2006). Geographically, our data sets pertain to almost all of northwest California. However, our fire severity data include few fires from the Cascade Range portion of the Shasta-Trinity and Klamath National Forests (eastern one-third of Fig. 1) or the coastal fog zone, because few fires in these areas grew to a size large enough to be assessed for fire severity by the MTBS program. Therefore, relationships and patterns that we found may not be representative of those areas. Additionally, the steeply dissected mountainous topography and strong summertime temperature inversion pattern of our study area plays a major role in driving fire behavior. As a result, trends and levels of severity and the relationships between severity and local climate conditions that we document for our study area may not hold true in other regions. For example, the percentage of high-severity fire in Sierra Nevada fires larger than 2000 ha occurring from 2000 through 2008 was more than twice as high as fires of the same size over the same period in our study area (32% vs. 15%, respectively).

The question remains as to how much climate change, changes in forest structure due to past management, and changes in fire suppression practices have influenced trends in fire severity and fire occurrence. While no trend in fire severity could be detected, our data suggest that fire frequency, size, and total burned area have strongly increased over the last 20+ years, and that climate is associated with a growing proportion of the variance in these variables. We believe that this pattern is the product of a changing climate plus increasing and more fire-prone fuels in some forest types, the latter driven by a combination of human- (e.g., fire suppression, land management practices) and climate-related (e.g., warming temperature, drier fire seasons) factors. Regardless, forested systems in northwestern California will burn under favorable weather conditions, and it is logical to expect more and larger fires under future climate change scenarios (Lenihan et al. 2008, State of California 2009, Gedalof 2011). A major question is whether we can influence the intensity at which future forest fires will burn, and thereby minimize the negative ecosystem effects of fire while maximizing the positive effects. Our results indicate that forest fires in northwestern California tend to be proportionally less severe in years when more total area burns over extended periods under a variety of burning conditions. This is likely because inversions that trap the large amount of smoke produced in such events reduce the probability of extreme fire behavior. Overall, the evidence suggests that, under the right meteorological, ecological, and political circumstances, wildland fires might be more extensively used in northwestern California to achieve management objectives such as reducing landscape-scale fire hazard, and restoring the ecological role of fire by increasing forest heterogeneity and sustaining biodiversity in fire-adapted forests. We recommend that managers consider these conclusions in developing fire management plans.

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Literature Cited

Agee, J. K. 1993. Fire ecology of Pacific Northwest forests. Island Press, Washington, D.C., USA.

Agee, J. K., and C. N. Skinner. 2005. Basic principles of forest fuel reduction treatments. Forest Ecology and Management 211:83–96.

Alexander, J. D., N. E. Seavy, C. J. Ralph, and B. Hogoboom. 2006. Vegetation and topographical correlates of fire severity from two fires in the Klamath-Siskiyou region of Oregon and California. International Journal of Wildland Fire 15:237–245.

Alley, W. M. 1984. The Palmer Drought Severity Index: limitations and assumptions. Journal of Climate and Applied Meteorology 23:1100–1109.

Arno, S. F., and S. Allison-Bunnell. 2002. Flames in our forest: disaster or renewal? Island Press, Washington, D.C., USA.

Arno, S. F., and C. E. Fiedler. 2005. Mimicking nature's fire: restoring fire-prone forests in the West. Island Press, Washington, D.C., USA.

Biswell, H. H. 1989. Prescribed burning in California wildlands vegetation management. University of California Press, Berkeley, California, USA.

Bond, W. J., and B. W. van Wilgen. 1996. Fire and plants. Chapman and Hall, London, UK.

Bowman, D. M. J. S., et al. 2009. Fire in the Earth system. Science 324:481–484.

Brohman, R., and L. Bryant, editors. 2005. Existing vegetation classification and mapping technical guide. General Technical Report WO-67. USDA Forest Service, Washington Office, Ecosystem Management Coordination Staff, Washington, D.C., USA.

Brown, T. J., B. L. Hall, and A. L. Westerling. 2004. The impact of twenty-first century climate change on wildland fire danger in the western United States: an applications perspective. Climatic Change 62:365–388.

CDF [California Department of Forestry and Fire Protection]. 2009. California statewide fire history 1906-2008. California Department of Forestry and Fire Protection, Sacramento, California, USA.

Chang, C. 1999. Understanding fire regimes. Dissertation. Duke University, Durham, North Carolina, USA.

Collins, B. M., J. D. Miller, A. E. Thode, M. Kelly, J. W. van Wagtendonk, and S. L. Stephens. 2009. Interactions among wildland fires in a long-established Sierra Nevada natural fire area. Ecosystems 12:114–128.

Collins, B., P. N. Omi, and P. L. Chapman. 2006. Regional relationships between climate and wildfire-burned area in the Interior West, USA. Canadian Journal of Forest Research 36:699–709.

Daly, C., R. P. Neilson, and D. L. Phillips. 1994. A statistical-topographic model for mapping climatological precipitation over mountainous terrain. Journal of Applied Meteorology 33:140–158.

Edwards, D., and B. C. Coull. 1987. Autoregressive trend analysis: an example using long-term ecological data. Oikos 50:95–102.

Eidenshink, J., B. Schwind, K. Brewer, Z.-L. Zhu, B. Quayle, and S. Howard. 2007. A project for monitoring trends in burn severity. Fire Ecology 3(1):3–21.

Field, C. B., G. C. Daily, F. W. Davis, S. Gaines, P. A. Matson, J. Melack, and N. L. Miller. 1999. Confronting climate change in California; ecological impacts on the Golden State. Union of Concerned Scientists, Cambridge, Massachusetts, USA; and Ecological Society of America, Washington, D.C., USA.

Franklin, J., C. E. Woodcock, and R. Warbington. 2000. Multi-attribute vegetation maps of forest service lands in California supporting resource management decisions. Photogrammetric Engineering and Remote Sensing 66:1209–1217.

Gedalof, Z. 2011. Climate and spatial patterns of wildfire in North America. Pages 89–116 *in* D. McKenzie, C. Miller, and D. A. Falk, editors. The landscape ecology of fire. Ecological Studies Volume 213. Springer-Verlag, New York, New York, USA.

Haight, R. G., and J. S. Fried. 2007. Deploying wildland fire suppression resources with a scenario-based standard response model. INFOR: Information Systems and Operational Research 45:31–39.

Heinselman, M. L. 1973. Fire in the virgin forests of the Boundary Waters Canoe Area, Minnesota. Quaternary Research 3:329–382.

Herweijer, C., R. Seager, E. R. Cook, and J. Emile-Gray. 2007. North American droughts of the last millennium from a gridded network of tree-ring data. Journal of Climate 20:1353–1376.

Holden, Z. A., P. Morgan, and A. T. Hudak. 2010. Burn severity of areas reburned by wildfires in the Gila National Forest, New Mexico, USA. Fire Ecology 6(3):77–85.

Husari, S., H. T. Nichols, N. G. Sugihara, and S. L. Stephens. 2006. Fire and fuel management. Pages 444–465 *in* N. G.

Sugihara, J. W. Van Wagtendonk, J. Fites-Kaufman, K. E. Shaffer, and A. E. Thode, editors. Fire in California's ecosystems. University of California Press, Berkeley, California, USA.

Jimerson, T. M., and D. W. Jones. 2003. Megram: blowdown, wildfire, and the effects of fuel treatment. Pages 55–59 *in* K. E. M. Galley, R. C. Klinger, and N. G. Sugihara, editors. Fire Conference 2000: The first national congress on fire ecology, prevention, and management. Tall Timbers Research Station. Tallahassee, Florida, USA.

Keeley, J. E. 2009. Fire intensity, fire severity and burn severity: a brief review and suggested usage. International Journal of Wildland Fire 18:116–126.

Key, C. H. 2006. Ecological and sampling constraints on defining landscape fire severity. Fire Ecology 2(2):34–59.

Key, C. H., and N. C. Benson. 2005*a*. Landscape assessment: remote sensing of severity, the Normalized Burn Ratio. Pages LA8–LA15 *in* D. C. Lutes, editor. FIREMON: Fire effects monitoring and inventory system. General technical Report RMRS-GTR-164-CD. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado, USA.

Key, C. H., and N. C. Benson. 2005*b*. Landscape assessment: remote sensing of severity, the Normalized Burn Ratio. Pages LA25–LA41 *in* D. C. Lutes, editor. FIREMON: Fire effects monitoring and inventory system. General technical Report RMRS-GTR-164-CD. USDA Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado, USA.

Kramer, C. Y. 1956. Extension of multiple range tests to group means with unequal number of replications. Biometrics 12:307–310.

Lenihan, J. H., D. Bachelet, R. P. Neilson, and R. Drapek. 2008. Response of vegetation distribution, ecosystem productivity, and fire to climate change scenarios for California. Climatic Change 87 (Supplement 1):S215–S230.

Leonzo, C. M., and C. R. Keyes. 2010. Fire-excluded relict forests in the Southeastern Klamath Mountains, California, USA. Fire Ecology 6(3):62–76.

Littell, J. S., D. McKenzie, D. L. Peterson, and A. L. Westerling. 2009. Climate and wildfire area burned in western U.S. ecoprovinces, 1916–2003. Ecological Applications 19:1003–1021.

Lutz, J., J. van Wagtendonk, A. Thode, J. Miller, and J. Franklin. 2009. Climate, lightning ignitions, and fire severity in Yosemite National Park, California, USA. International Journal of Wildland Fire 18:765–774.

Marlon, J. R., et al. 2009. Wildfire responses to abrupt climate change in North America. Proceedings of the National Academy of Sciences USA 106:2519–2524.

McKelvey, K. S., and K. K. Busse. 1996. Twentieth Century Fire Patterns on Forest Service Lands. Pages 1119–1138 *in* Sierra Nevada Ecosystems Project: final report to Congress. University of California, Davis, California, USA.

McKelvey, K. S., C. N. Skinner, C. Chang, D. C. Erman, S. J. Hussari, D. J. Parsons, J. W. van Wagtendonk, and C. P. Weatherspoon. 1996. An overview of fire in the Sierra Nevada. Pages 1033–1040 *in* Sierra Nevada Ecosystems Project: final report to Congress. University of California, Davis, California, USA.

McKenzie, D., Z. E. Gedalof, D. L. Peterson, and P. Mote. 2004. Climatic change, wildfire, and conservation. Conservation Biology 18:890–902.

Meyn, A., S. W. Taylor, M. D. Flannigan, K. Thonicke, and W. Cramer. 2010. Relationship between fire, climate oscillations, and drought in British Columbia, Canada, 1920–2000. Global Change Biology 16:977–989.

Miller, C. 2003. Simulation of effects of climatic change on fire regimes. Pages 69–94 *in* T. T. Veblen, W. L. Baker, G. Montenegro, and T. W. Swetnam, editors. Fire and climatic change in temperate ecosystems of the western Americas. Springer-Verlag, New York, New York, USA.

Miller, J. D., E. E. Knapp, C. H. Key, C. N. Skinner, C. J. Isbell, R. M. Creasy, and J. W. Sherlock. 2009*a*. Calibration and validation of the relative differenced Normalized Burn Ratio (RdNBR) to three measures of fire severity in the Sierra Nevada and Klamath Mountains, California, USA. Remote Sensing of Environment 113:645–656.

Miller, J. D., H. D. Safford, M. A. Crimmins, and A. E. Thode. 2009*b*. Quantitative evidence for increasing forest fire severity in the Sierra Nevada and southern Cascade Mountains, California and Nevada, USA. Ecosystems 12:16–32.

Miller, J. D., and A. E. Thode. 2007. Quantifying burn severity in a heterogeneous landscape with a relative version of the delta Normalized Burn Ratio (dNBR). Remote Sensing of Environment 109:66–80.

Moran, M. D. 2003. Arguments for rejecting the sequential Bonferroni in ecological studies. Oikos 100:403–405.

Morgan, P., E. K. Heyerdahl, and C. E. Gibson. 2008. Multi-season climate synchronized forest fires throughout the 20th century, Northern Rockies, USA. Ecology 89:717–728.

NOAA. 2009*a*. Palmer Drought Severity Index for California Climate Division 2. National Oceanic and Atmospheric Administration (NOAA) National Climate Data Center, Asheville, North Carolina, USA. http://www1.ncdc.noaa.gov/pub/data/cirs/

NOAA. 2009*b*. Northern Hemisphere Teleconnection Indices. National Oceanic and Atmospheric Administration (NOAA) Climate Prediction Center, Asheville, North Carolina, USA. ftp://ftp.cpc.ncep.noaa.gov/wd52dg/data/indices/tele_index.nh

Noss, R. F., J. F. Franklin, W. L. Baker, T. Schoennagel, and P. B. Moyle. 2006. Managing fire-prone forests in the western United States. Frontiers in Ecology and the Environment 4:481–487.

Odion, D. C., E. J. Frost, J. R. Strittholt, H. Jiang, D. A. Dellasala, and M. A. Moritz. 2004. Patterns of fire severity and forest conditions in the Western Klamath Mountains, California. Conservation Biology 18:927–936.

Odion, D. C., M. A. Moritz, and D. A. Dellasala. 2010. Alternative community states maintained by fire in the Klamath Mountains, USA. Journal of Ecology 98:96–105.

Pausas, J. G. 2004. Changes in fire and climate in the eastern Iberian Peninsula (Mediterranean Basin). Climatic Change 63:337–350.

Power, M., et al. 2008. Changes in fire regimes since the Last Glacial Maximum: an assessment based on a global synthesis and analysis of charcoal data. Climate Dynamics. 30:887–907.

Pyne, S. J., P. L. Andrews, and R. D. Laven. 1996. Introduction to wildland fire. Second edition. John Wiley and Sons, New York, New York, USA.

Robock, A. 1988. Enhancement of surface cooling due to forest fire smoke. Science 242:911–913.

Robock, A. 1991. Surface cooling due to forest fire smoke. Journal of Geophysical Research 98(D11) 20:869–878.

Rorig, M. L., and S. A. Ferguson. 1999. Characteristics of lightning and wildland fire ignition in the Pacific Northwest. Journal of Applied Meteorology 38:1565–1575.

Rorig, M. L., and S. A. Ferguson. 2002. The 2000 fire season: lightning-caused fires. Journal of Applied Meteorology 41:786–791.

Rorig, M. L., S. J. McKay, S. A. Ferguson, and P. Werth. 2007. Model-generated predictions of dry thunderstorm potential. Journal of Applied Meteorology and Climatology 46:605–614.

Rothman, K. J. 1990. No adjustments are needed for multiple comparisons. Epidemiology 1:43–46.

Running, S. W. 2006. Is global warming causing more, larger wildfires? Science 313:927–928.

Sawyer, J. O. 2007. Forests of northwestern California. Pages 253–295 *in* M. G. Barbour, T. Keeler-Wolf, and A. A. Schoenherr, editors. Terrestrial vegetation of California. University of California Press, Berkeley, California, USA.

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

Schowengerdt, R. A. 1997. Remote sensing: models and methods for image processing. Second edition. Academic Press, New York, New York, USA.

Shumway, R. H. 1988. Applied statistical times series analysis. Prentice Hall, Englewood Cliffs, New Jersey, USA.

Skinner, C. N., A. H. Taylor, and J. K. Agee. 2006. Klamath Mountains bioregion. Pages 170–194 *in* N. G. Sugihara, J. W. Van Wagtendonk, J. A. Fites-Kaufman, K. E. Shaffer, and A. E. Thode, editors. Fire in California ecosystems. University of California, Berkeley, California, USA.

State of California. 2009. 2009 California climate adaptation strategy. Report CNRA-1000-2009-027. California Natural Resources Agency, Sacramento, California, USA. http://www.energy.ca.gov/2009publications/CNRA-1000-2009-027/CNRA-1000-2009-027-F.PDF

Stephens, S. L. 2005. Forest fire causes and extent on United States Forest Service lands. International Journal of Wildland Fire 14:213–222.

Stephens, S. L., R. E. Martin, and N. E. Clinton. 2007. Prehistoric fire area and emissions from California's forests, woodlands, shrublands and grasslands. Forest Ecology and Management 251:205–216.

Sugihara, N. G., J. W. van Wagtendonk, K. E. Shaffer, J. Fites-Kaufman, and A. E. Thode, editors. 2006. Fire in California's ecosystems. University of California Press, Berkeley, California, USA.

Swetnam, T. W. 1993. Fire history and climate change in giant sequoia groves. Science 262:885–889.

Taylor, A. H., and C. N. Skinner. 1998. Fire history and landscape dynamics in a late-successional reserve, Klamath Mountains, California, USA. Forest Ecology and Management 111:285–301.

Taylor, A. H., and C. N. Skinner. 2003. Spatial patterns and controls on historical fire regimes and forest structure in the Klamath Mountains. Ecological Applications 13:704–719.

Thompson, J. R., T. A. Spies, and L. M. Ganio. 2007. Reburn severity in managed and unmanaged vegetation in a large wildfire. Proceedings of the National Academy of Science USA 104:10743–10748.

Trouet, V., A. H. Taylor, A. M. Carleton, and C. N. Skinner. 2006. Fire-climate interactions in forests of the American Pacific coast. Geophysical Research Letters 33:L18704.

Trouet, V., A. H. Taylor, E. R. Wahl, C. N. Skinner, and S. L. Stephens. 2010. Fire-climate interactions in the American West since 1400 CE. Geophysical Research Letters 37:L04702.

USDA. 2005. CALVEG zones and alliances: vegetation descriptions. USDA Forest Service, Pacific Southwest Region, Remote Sensing Lab, Sacramento, California, USA. http://www.fs.fed.us/r5/rsl/projects/classification/

USDA. 2006. Wildland urban intermix. USDA Forest Service, Pacific Southwest Region, Remote Sensing Lab, Sacramento, California, USA. http://www.fs.fed.us/r5/rsl/clearinghouse/gis-download.shtml

Wallace, J. M., and D. S. Gutzler. 1981. Teleconnections in the geopotential height field during the Northern Hemisphere winter. Monthly Weather Review 109:784–812.

Weatherspoon, C. P., and C. N. Skinner. 1995. An assessment of factors associated with damage to tree crowns from the 1987 wildfires in northern California. Forest Science 41:430–451.

Weatherspoon, C. P., and C. N. Skinner. 1996. Landscape-level strategies for forest fuel management. Pages 1471–1492 in Sierra Nevada Ecosystem Project: final report to Congress. University of California, Davis, California, USA.

Westerling, A. L., H. G. Hidalgo, D. R. Craven, and T. W. Swetnam. 2006. Warming and earlier spring increase Western U.S. forest wildfire activity. Science 313:940–943.

Whitlock, C., J. Marlon, C. Briles, A. Brunelle, C. Long, and P. Bartlein. 2008. Long-term relations among fire, fuel, and climate in the north-western US based on lake-sediment studies. International Journal of Wildland Fire 17:72–83.

Williams, A. A., D. J. Karoly, and N. Tapper. 2001. The sensitivity of Australian fire danger to climate change. Climatic Change 49:171–191.

Wills, R. D., and J. D. Stuart. 1994. Fire history and stand development of a Douglas-fir hardwood forest in Northern California. Northwest Science 68:205–212.

WRCC [Western Regional Climate Center]. 2009. California climate tracker. WRCC, Reno, Nevada, USA. http://www.wrcc.dri.edu/monitor/cal-mon/index.html

Yin, X. 1999. The decay of forest woody debris: numerical modeling and implications based on some 300 data cases from North America. Oecologia 121:81–98.

Supplemental Material

Appendix

Methods used to create a 1987 prefire vegetation map and accuracy assessment results (*Ecological Archives* A022-011-A1).