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Quantitative Evidence for Increasing Forest Fire Severity in the Sierra Nevada and Southern Cascade Mountains, California and Nevada, USA

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Abstract

Recent research has concluded that forest wildfires in the western United States are becoming larger and more frequent. A more significant question may be whether the ecosystem impacts of wildfire are also increasing. We show that a large area (approximately 120000 km^2) of California and western Nevada experienced a notable increase in the extent of forest stand-replacing (''high severity'') fire between 1984 and 2006. High severity forest fire is closely linked to forest fragmentation, wildlife habitat availability, erosion rates and sedimentation, post-fire seedling recruitment, carbon sequestration, and various other ecosystem properties and processes. Mean and maximum fire size, and the area burned annually have also all risen substantially since the beginning of the 1980s, and are now at or above values from the decades preceding the 1940s, when fire suppression became national policy. These trends are occurring in concert with a regional rise in temperature and a long-term increase in annual precipitation. A close examination of the climate–fire relationship and other evidence suggests that forest fuels are no longer limiting fire occurrence and behavior across much of the study region. We conclude that current trends in forest fire severity necessitate a reexamination of the implications of all-out fire suppression and its ecological impacts.

Key words: California; fire ecology; fire severity; burn severity; Sierra Nevada; Sierra Nevada Forest Plan Amendment; relative differenced Normalized Burn Ratio.

Introduction

The fundamental discord between human desires for economic and ecological stability, and the disturbance-prone nature of many western ecosystems has become a defining feature of the American West, and the dissonance is rising. Recent studies have shown that wildfire frequency,

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size, and overall burned area per annum are all increasing across much of the western United States (McKelvey and others 1996; Stephens 2005; Westerling and others 2006), and home losses and wildfire-related human fatalities are up as well (Torn and others 1999; CDF 2007b). Ironically, these alarming trends occur against a background of massive fire-fighting intervention (''fire suppression''), which has reduced forest fire activity to levels far below those that characterized many western landscapes before Euroamerican settlement began in the mid-19th century (Agee 1993; Sugihara and others 2006; Stephens and others 2007). This artificial condition has set the stage for a century of economic development, population growth, and land management policy-making that is in marked disequilibrium with the underlying ecological template across much of the western US. Given dependence of the human status quo on the maintenance of relatively fire-free conditions, recent increases in fire activity have been treated with alarm, and expenditures for enhanced levels of fire suppression are rising rapidly (WFLC 2004).

Although concerns for human safety and property may be well founded, the ecological consequences of increasing fire activity are less certain. Many western American forest ecosystems are adapted to frequent wildfire, numerous western plant and animal species evolved in close linkage to fire, and the nature of many fundamental ecosystem processes has been dramatically altered by fire's absence (Arno and Fiedler 2005; Noss and others 2006; Sugihara and others 2006). Indeed, it is an often-repeated mantra that the major ecological issue facing western forests today is the relative *absence* of fire. A major public debate has emerged, pitting those who would act decisively to halt recent trends in fire activity against those who would step back and ''let nature take its course'', however adversaries in the debate have colored the problem black and white, and battle lines have been drawn before all of the information is in.

Recent research is helping to illuminate the variegated nature of the ''western fire problem''. For example, we now know that different ecosystem types and geographic areas are naturally characterized by different fire regimes, and that in some places current fire activity is not at all abnormal (for example, in many moist, higher elevation and/or higher latitude forests historically characterized by infrequent, highly severe fires), whereas in others it is far outside the historic range of variability (for example, in many dry, lower elevation pine-dominated forests historically characterized by high frequency, low severity fire regimes) (Agee 1993; Allen and others 2002; Schoennagel and others 2004; Arno and Fiedler 2005; DeWilde and Chapin 2006; Noss and others 2006). We have also learned that the relative roles of fuels and climate in driving fire activity are geographically and temporally variable, and depend on factors ranging from local vegetation and topography to global-scale interactions between ocean and atmosphere (Heyerdahl and others 2001; Westerling and Swetnam 2003; Schoennagel and others 2004). Thus, even if fire activity were increasing at equal rates across the West (and it is not), it might be for different reasons and it might have different ecological consequences depending on the place and context.

What is thus far missing in the debate about increased wildfire activity in the West is a rigorous consideration of the actual ecosystem impacts, or ''severity'' of fire. In much of the West, forest wildfires may be becoming larger and more frequent, but until now there has been little compelling quantitative support for the idea that fires are becoming generally more ''catastrophic'', outside of anecdote, theoretical considerations, and a handful of local studies (Skinner and Chang 1996; Gruell 2001; USDA 2004; Holden and others 2007). In this contribution, we report results from a long-term, broad-scale assessment of patterns in the extent of high severity (that is, forest stand-replacing) fire in a 750 × 160 km area of California and western Nevada, centered on the Sierra Nevada and southern Cascade Mountains. We used a relativized version of the delta Normalized Burn Ratio (NBR) (Miller and Thode 2007) calculated from 1 year pre- and 1 year post-fire Landsat images and calibrated with extensive field data, to assess fire severity in 202 wildfires larger than 40 ha in size that burned 490,277 ha between 1984–2006 in the study region. We stratified our data by forest type, and measured temporal trends across the study period in severity, heterogeneity (''patchiness'') of high severity fire, and area burned. We also used a 99-year dataset (1908–2006) of medium and large (>40 ha) fire perimeters from 3171 fires in the study region to evaluate longer-term trends in fire number, fire size, and annual area burned. Finally, we assessed the role of a suite of macroclimatic variables in explaining trends in both datasets.

Material and Methods

Spatial Extent and Time Period

The approximately 120,000 km^2 study region is formed by the analysis area of the Sierra Nevada

Forest Plan Amendment (SNFPA) (USDA 2004), which guides land and resource management on 50000 km^2 of National Forest land on 11 US National Forests (Figure 1). Vegetation in the study region is dominated by conifer forest; forest types that occur in the fires sampled for this project are listed in Table 1. Climate is Mediterranean-type, with warm, dry summers and cool, wet winters; almost all precipitation falls between October and April. Elevations in the study region range from approximately 300 m along the western border to over 4000 m along the Sierra Nevada crest. Geographic centroids of fires sampled for severity ranged from 420 m elevation to 2853 m.

For our severity analyses, we sampled fires that occurred at least partially on Forest Service managed lands, independent of the place of fire origin. In total 72% of the sampled fire area occurs on Forest Service (USFS) lands, 28% on other ownerships (Table 2). Note that almost all non-USFS areas were owned or managed by private owners, the State of California or the Bureau of Land Management, and are subject to the same fire suppression policies as most USFS lands. We only analyzed fires occurring at least partially on Forest Service lands because the purpose of the monitoring project we report on was primarily to ascertain severity patterns in forest fires on the National Forests in the SNFPA area, and to test basic assumptions made in the SNFPA itself regarding severity patterns in forestlands managed by the Forest Service. Most of our analyses apply to the entire study region, except for analyses stratified by vegetation type, which apply only to USFS lands. The vegetation stratification needed to be based on a standardized and frequently updated vegetation

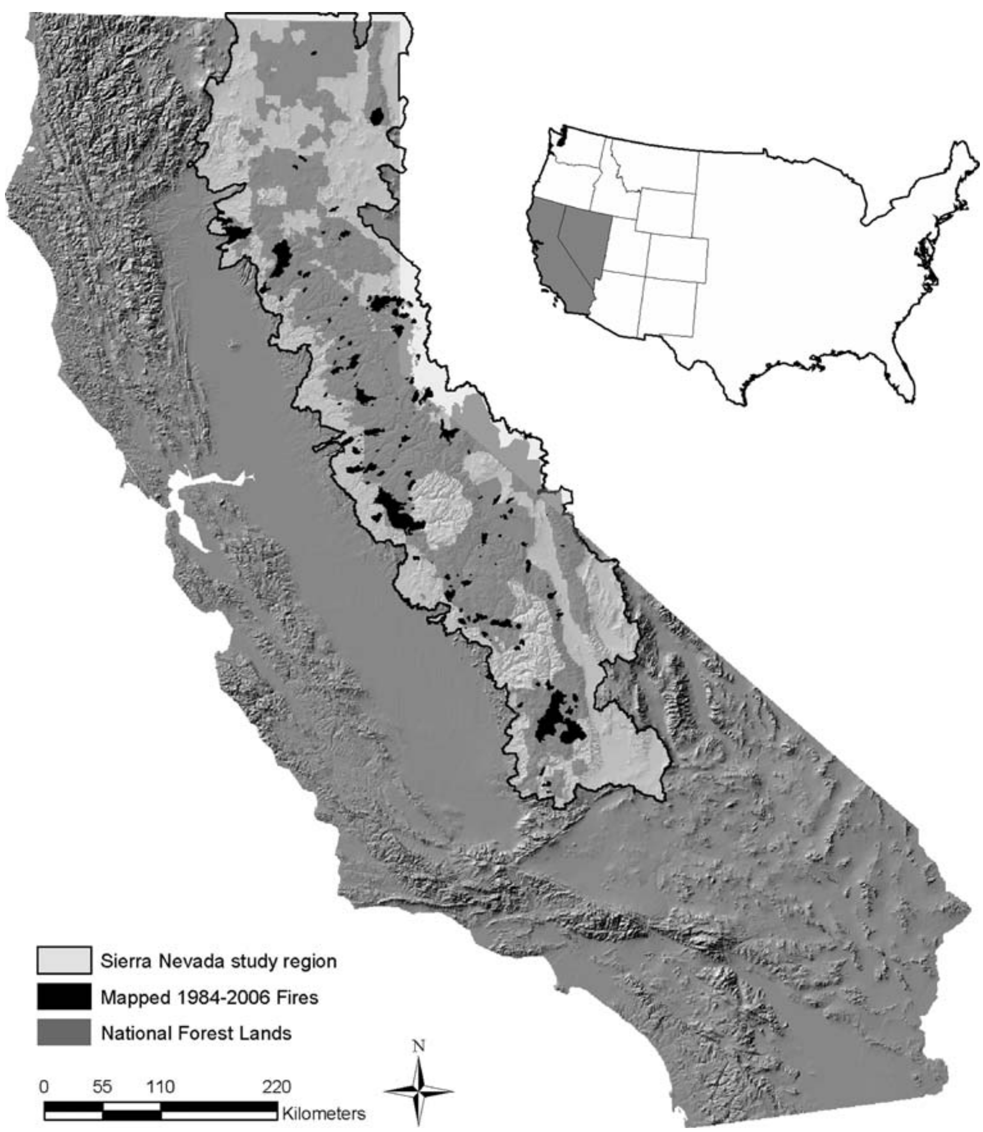

Figure 1. Map of the overall study region and the Sierra Nevada Forest Plan Amendment (SNFPA) area in California and western Nevada. Within the study region, areas demarcated in gray are National Forest lands. Black polygons within the study area represent 202 fires analyzed in the current study.

Table 1. Area Mapped by Vegetation Type in 177 Fires that Occurred During 1984–2004 on Forest Service Lands, Plus Percent of Total Area with High Severity Fire Effects in Each Forest Type

Vegetation type	Area (ha)	% High severity
Black oak	5918	23
Blue oak	7537	2
Eastside pine	23439	40
Live oak	21579	18
Lodgepole pine	2977	9
Mixed conifer	97252	29
Ponderosa pine	24709	26
Red fir	12080	13
Riparian	1438	15
Subalpine conifer	1754	7
White fir	9653	33

Results for chaparral shrub (64240 ha) and pinyon juniper (15894 ha) types are not reported in this article.

Table 2. Fire Area Sampled in the Fire Severity Study, 1984–2006, by Land Ownership

Ownership	Area (ha)
Private lands	97011
State lands	11061
BLM	23418
NPS	9671
USFS	349116

map, which only existed for Forest Service lands, thus our analyses based on specific forest types were *per force* restricted to the National Forests. For analysis not stratified by vegetation type we report on patterns found across the entire dataset because (1) the general patterns we found were congruent across ownership boundaries, and (2) we were interested in documenting patterns in fire across the study region as a whole. Five percent of the sampled fires (3.1% of sampled fire area) were managed for Wildland Fire Use (WFU), which allows naturally ignited fires to burn under some circumstances; these percentages are comparable to the overall values in the State fire history database (see below).

Our sample of 202 fires included about 60% of the total area burned from 1984 to 2006 in the study region. Our sample included all but five fires larger than 400 ha in area that occurred in the study region from 2000 to 2006, all fires larger than 40 ha in area that occurred in the central study region 1984–2004, and a subset of fires larger than 40 ha that occurred elsewhere in the study region. In addition, we calculated severity values for all fires larger than 400 ha occurring in 2007 (53660 ha total) using the first clear post-fire Landsat image. We did not use the 2007 fires in statistical analyses because we do not yet have 1-year post-fire images. Considering fires larger than 40 ha, the size-distribution of our sample is not different from the full fire record for the study region from 1984 to 2006 ($\chi^2 = 9.051$, $P = 0.249$, df = 7), and mean fire sizes of our sample and the study region are comparable ($t = 1.172$, $P = 0.242$, df = 614). There is no latitudinal gradient in the severity of fires smaller than 400 ha in size ($R^2 < 0.000$), that is, there is no difference in small fire severity between the north, central, and south study region that might bias our results. Areas mapped as shrubland were removed from all fire areas before analysis.

About 1,100,000 ha of USFS lands in the study area are managed as WFU areas, where naturally ignited fires may be allowed to burn without direct suppression, depending on a variety of factors including weather, fire location, staffing, and budget. In addition, about 80% (*c.* 500,000 ha) of Yosemite and Sequoia-King Canyon National Parks are managed as WFU. Across the broader study region, about 13% of all lands are managed under WFU authority, but because these areas are in almost all cases high elevation wilderness areas with low fuel loads, WFU fires tend to remain small. Because relatively few fires are managed as WFU, the total area burned in WFU fires is very small as well. The first WFU fire in the State fire history database occurred in 1970. Since then 120 WFU fires have been captured in the database, totaling 49705 ha and averaging 414 ha (versus 902 ha mean for all fires). Considering only fires larger than 40 ha in size, WFU fires are 6.5% of the fires in the database since 1970 (72/1113), and 3.9% of the area (49013 ha/1256456 ha). Coverage of WFU fires in our severity dataset is 5% of sampled fires (10/202) and 3.1% of sample area (15079 ha/490277 ha). Finally, our 1908–2006 dataset of fire occurrence and size includes all fires over 40 ha in the study region, independent of management scenario. We conclude that our coverage of WFU fires is approximately proportional to their occurrence in the study region, and the patterns we report are therefore representative of the study region as a whole and are not biased toward or against areas managed for WFU.

There were no temporal trends in precipitation from 1984 to 2006. Based on regional data from WRCC (2007), the 1984–2006 period was at the 112-year mean for precipitation (mean annual precipitation from 1984 to 2006 divided by 112-

year mean = 1.002) and precipitation variability (average of annual standard deviations from 112-year mean = 0.327; 1984–2006 mean of standard deviations = 0.326). Mean maximum temperatures did not change significantly from 1984 to 2006; with respect to the mean minima, only the mean for June–August increased (+0.9° C, R^2 = 0.177, P = 0.046).

Image Preprocessing

Fire severity was mapped with Landsat TM satellite imagery, using the NBR as the basis for our severity measures (Key and Benson 2005b). To allow interfire comparisons of fire severity and to remove the biasing effect of the pre-fire condition, we used a relativized version of dNBR (''RdNBR''), dividing the dNBR image by a function of the prefire NBR (Miller and Thode 2007; Safford and others 2008). Landsat images were converted to ''at-sensor-reflectance'' as described by Chander and Markham (2003) before calculating RdNBR. All post-fire images were acquired one growing season after fire occurrence to match the date of field sampling (Key and Benson 2005a). Environmental factors such as atmospheric conditions, topography, surface moisture, seasonal phenology, and solar zenith angle all influence analysis of multitemporal data in a change detection protocol (Coops and others 2006). We minimized seasonal phenology, surface moisture, and solar zenith angle differences by matching pre- and post-fire image dates as closely as possible (Singh 1989; Coppin and Bauer 1996). Images from June through August were used to map 96% of the fires. All images were geometrically registered using terrain correction algorithms (level 1T) by the USGS-EROS Data Center. No atmospheric scattering algorithm was applied to the data because the NBR employs only near and middle infrared wavelengths that are minimally affected by atmospheric scattering, especially during the summer (Avery and Berlin 1992); our study region has a Mediterranean climate and practically no summertime rainfall. We were only interested in analyzing deforestation due to stand-replacing fire, and previous projects have successfully mapped burned areas with at least 50% canopy mortality using Landsat NIR and SWIR bands without atmospheric correction (for example, Fraser and others 2004). Temporal radiometric differences are minimal relative to the change in reflectance due to stand-replacing deforestation (Cohen and others 1998), and atmospheric correction methods can produce higher Root Mean Square Error (RMSE) values across multiple dates than correction only to at-sensor-reflectance (Schroeder and others 2006). Satellite values were not corrected for topographic shading because NBR is a ratio and topographic effects cancel when atmospheric scattering is minimal (Kowalik and others 1983; Ekstrand 1996). NBR values were multiplied by 1000 and converted to integer format to match procedures established by Key and Benson (2005b).

To allow fire severity comparisons between fires across space and time, we carried out two data normalizations. First, to account for imagery calibration errors and inter-annual differences in phenology, we subtracted the average dNBR value from unburned areas outside the fire perimeter from the within-fire dNBR values. This normalization process sets the unburned/unchanged condition to zero. Second, we normalized the dNBR absolute difference image by dividing by a function of the pre-fire image to create a relative index (RdNBR; Miller and Thode 2007). In this fashion, pixel values were converted to a ratio that theoretically ranges between zero percent for unburned areas and 100% for areas with total aboveground vegetation mortality (note that, in practice, RdNBR can increase in value where complete vegetation mortality has occurred, as the dNBR index is also sensitive to ash, char, and substrate composition [Kokaly and others 2007]). In sum, RdNBR is both spatially and temporally normalized, which allows for the development of thresholds from one set of fires to be applied to other fires across time and space (Miller and Thode 2007).

Field Calibration and Validation

To field calibrate the Landsat data, we sampled 1122 plots on 18 fires 1-year postfire across the study region between 2002 and 2005. Field plots were circular, with a radius of 45 m in 2002, 2003 and 2004, and a radius of 30 m in 2005. Our analysis assumes that 4 years of field-collected data are sufficient to adequately calibrate two decades of remotely sensed fire severity data; our accuracy assessments (below) suggest that this is a reasonable assumption. Plots were sampled in approximate proportion to the total areal extent of the vegetation type in question (see Table 1). Quantitative data on tree survival were collected, as well as a semi-quantitative assessment of overall severity to vegetation, which we made using the Composite Burn Index (CBI) protocol (Key and Benson 2005a). In this article we measure fire severity in units of the CBI. The CBI is an integrated field-based measure of fire effects based principally on an ocular assessment of the quantity of fuel consumed and the extent of vegetation regeneration. Only one post-fire visit is made to each plot, and a number of variables are estimated relative to their prefire condition. Eighteen vegetation related variables are assessed (divided into four vertical strata: herbaceous and low shrubs, tall shrubs and small trees, intermediate trees, and dominant and codominant trees), as well as four surface fuel variables, and one soil variable (Key and Benson 2005a). Because the CBI protocol calls for field sampling one-year postfire, our definition incorporates both immediate and one-year postfire effects (Lentile and others 2006), and thus may conflate the separate contributions of fire severity and ecosystem response (Keeley 2008). The field data were used to calibrate the satellite derived RdNBR index to the CBI severity measure (ranging from 0 [unburned] to 3 [highest severity]), then the data were divided into low (which includes unchanged), moderate and high severity classes. The low-moderate severity threshold was set at CBI = 1.25 and the moderate-high severity threshold at CBI = 2.25 (see Miller and Thode [2007] for details on selection of CBI thresholds and regression modeling of RdNBR to CBI).

We assessed accuracy of the high severity class in our severity maps in two ways. First, we calculated classification accuracies using cross-validation, which calculates accuracy from the classified data themselves. The producer's accuracy from the standard confusion matrix for all forested vegetation types (≥10% tree cover, as per standard USFS practice [Brohman and Bryant 2005]) was 80.2%, user's accuracy was 76.2% (see Miller and Thode [2007] for a cross-validation accuracy assessment including shrub types). Second, we calculated classification accuracies using completely independent plot data from six large wildfires in Yosemite National Park (Zhu and others 2006): producer's accuracy was 70.7% and user's accuracy 84.3% (Table 3). These values are consistent with those reported by other researchers (Cocke and others 2005; Epting and others 2005; Stow and others 2007). In addition to measuring CBI, we collected quantitative data on tree mortality (live and dead by species and size class), and calculated pre- and post-fire canopy cover using the Forest Vegetation Simulator (FVS) (Dixon 2002). Regression of our CBI values to percent change in tree canopy cover ($R^2 = 0.56$, $P < 0.0001$) indicates that at CBI values of 2.25 (our high severity threshold) and above more than 90% of the tree canopy had been directly consumed by fire or scorched by fire and died within the subsequent year (our field assessments were always conducted one-year postfire).

Vegetation Type Stratification

Our analyses by vegetation type only include USFS lands, as we lack conformable and regularly updated vegetation maps for the other ownerships. Vegetation types were based on the original SNFPA stratification (USDA 2004), with a few types added for completeness. We used the most recently available CALVEG vegetation map (USDA 2006) to stratify the area in each fire from 1984 to 2004 into forest types growing on the site before fire; see Table 1 for type names and areas.

Trends in Percent High Severity and Patch Size

We used time series regression to calculate trends in the percent of fire area burning at high severity per year 1984–2006. Because our vegetation type stratification only covered the period 1984–2004, fires from 2005 to 2006 were not included in analyses of specific forest types. The high severity patch size analysis only includes conifer types, excluding pinyon-juniper, 1984–2004. Minimum measurable patch size was 900 m^2, due to the Landsat 30 m pixel. All severity data were transformed by arcsin-square root and all area data by log to meet statistical assumptions of normality. We

Table 3. CBI Confusion Matrix for Plots from Six Fires in Yosemite National Park

Class name	Unchanged	Low	Moderate	High	Total	User's accuracy (%)
Unchanged	14	9	4		27	51.9
Low	5	43	49	4	101	42.6
Moderate		9	28	25	62	45.2
High			13	70	83	84.3
Total	19	61	94	99	273	
Producer's accuracy (%)	73.7	70.5	29.8	70.7		56.8

Includes non-forested plots; CBI data provided by Carl Key, reported in Zhu and others (2006).

fit Autoregressive Integrated Moving Average (ARIMA) time domain regressions to the severity data, using Box-Jenkins techniques for model identification and estimation; we compared model goodness-of-fit using the Akaike Information Criterion (Shumway 1988). Due to high inter-annual variability in the datasets, we also graphically portray trends using a moving average of the annual data. We chose a 10-year window for our moving average calculation because (1) temporal autocorrelation statistics among the fire severity data maximized at 10 years; (2) 10 years is approximately the length of a half-cycle of the Pacific Decadal Oscillation (PDO) (Mantua and Hare 2002), which has demonstrated temporal effects on fire activity in Northern California (Sugihara and others 2006); and (3) we wanted at least 10 points to track mean trends across the analysis period.

Trends in Area Burned and Fire Occurrence, 1908–2006

We used the interagency California digital fire history database (CDF 2007a) to examine trends in number of fires, fire size and annual burned area in the study region. This is the most comprehensive, long-term database of fire polygons in the western United States. It is considered more or less complete for fires larger than 4 ha back to 1950, and mostly comprehensive for fires before that period, back to about 1908, when the USFS began recording fires. Before that date, data quality deteriorates significantly. For that reason, 1908 has been the standard beginning date for a number of previous studies (for example, Erman and Jones 1996; McKelvey and Busse 1996). Except for fires smaller than 40 ha, there is no systematic exclusion of fires from the database that would bias an analysis of trends. A considerable amount of time has been spent validating and updating the database over the last decade by USFS and Department of Interior staff and it is currently in a more complete state than when it was originally used by McKelvey and Busse in 1996. This notwithstanding, the database is still subject to errors associated with human archival data (McKelvey and Busse 1996; Morgan and others 2001). In our analysis we included all fires larger than 40 ha that occurred within the study region from 1908 to 2006 (total $n = 3171$) because (1) smaller fires tend to be under-reported before 1950, and (2) fires larger than 40 ha tend to represent those that escape initial containment. We used 11-year centered running means of the dependent variables to depict long-term trends.

Climate Analysis

To examine broad-scale fire–climate relationships we acquired Sierra Nevada climate data summaries (WRCC 2007) for total precipitation, and mean minimum and maximum temperatures; monthly data were grouped into standard 3-month seasons (winter = Dec–Feb, and so on). Stepwise linear regressions for all subsets of the climate variables were carried out versus percent high severity by forest type, 1984–2004, and number of fires, fire size, and annual burned area, 1908–2006. Percent data were arcsin-square root transformed and area variables log transformed to meet statistical assumptions of normality; data collinearity was assessed using variance inflation factors. The time series was divided into three temporal groups to determine whether different climate variables were correlated to fire size and count in the early (1908–1956), late (1957–2006), and very late (1982–2006) portions of the study period; these temporal groups were generated by splitting the data set in half and then in half again, and were not based on *a priori* assumptions.

Trends in mean fire size (log transformed) between 1908 and 2006 were also analyzed in relation to two broad-scale climate drivers: the PDO, and the El Niño-Southern Oscillation (ENSO). Standardized PDO index values (annual means) were acquired from JISAO (2007); ENSO data were acquired as the NIÑO 3.4 Index (annual means), from KNMI (2007). We ''lagged'' the PDO and NIÑO 3.4 indices by 1–10 years against log fire size by moving the index values forward and backward with relation to the target year and calculating Pearson correlations under each lag. Overall correlations were strongest under the 5, 6, and 7-year lag conditions for PDO (PDO measured 5–7 years before the fire size target year), and an 8-year lag for NIÑO 3.4. Under these lag conditions, we also computed Pearson correlations between the PDO and NIÑO 3.4 and each of the running 20-year periods ($n = 80$) from 1908 to 2006.

Results

Temporal Trends in Stand-Replacing Fire, 1984–2006

Trends in the proportional area of wildfires burning at high severity showed strong inter-annual variability, but time-series regression documents significant increases in fire severity in forest types that make up the majority (~70%) of the burned area we surveyed (Table 4). At the beginning of the

Table 4. Regression Statistics for Results of ARIMA Time Series Modeling of Fire Severity and High Severity Patch Size, 1984–2004

	Percent high severity (arcsin-square root transformed)									Patch size (ha, log transformed)	
	Mixed conifer	Ponderosa pine	Eastside pine	Red fir	White fir	Black oak	Blue oak	Live oak	All forest 1984–2006	Mean	Mean maximum
n	20	19	17	14	14	16	15	18	23	19	19
Error degrees of freedom (dfe)	17	17	15	12	8	14	11	16	20	17	17
Parameter estimates											
Sigma-sq	0.017	0.022	0.035	0.138	0.016	0.092	0.008	0.010	0.008	0.116	0.468
Intercept	0.197	0.255	0.138	0.315	0.003	0.069	0.011	0.143	0.227	0.112	0.550
Linear	0.008	−0.001	0.010	0.007	0.019	0.013	0.013	0.008	0.006	0.036	0.077
Autoregressive function (AR) 1	−0.546	−0.611			−1.284		−1.198		−0.635		
AR2					−1.585		−0.755				
AR3					−1.174						
AR4					−0.786						
P (linear)	0.025	0.789	0.152	0.237	<0.0001	0.002	<0.0001	0.059	0.003	0.011	0.007
P (AR1)	0.017	0.006			0.000		<0.0001		0.001		
P (AR2)					0.001		0.002				
P (AR3)					0.003						
P (AR4)					0.001						
Statistics of fit											
Mean square error (MSE)	0.019	0.019	0.031	0.019	0.023	0.008	0.012	0.009	0.007	0.104	0.418
Root mean square error (RMSE)	0.016	0.139	0.175	0.138	0.151	0.092	0.109	0.094	0.083	0.322	0.647
Mean absolute % error (MAPE)	53.077	31.557	56.254	30.869	52.279	28.070	269.487	33.186	23.236	199.745	68.484
Mean absolute error (MAE)	0.106	0.111	0.158	0.095	0.108	0.070	0.093	0.076	0.067	0.273	0.523
R^2	0.356	0.252	0.158	0.114	0.462	0.522	0.249	0.214	0.465	0.325	0.357
adj R^2	0.281	0.158	0.074	0.040	0.126	0.488	0.044	0.165	0.411	0.286	0.319
Akaike information criterion	−77.194	−68.883	−55.190	−53.694	−40.948	−74.468	−58.466	−80.993	−108.499	−39.073	−12.559

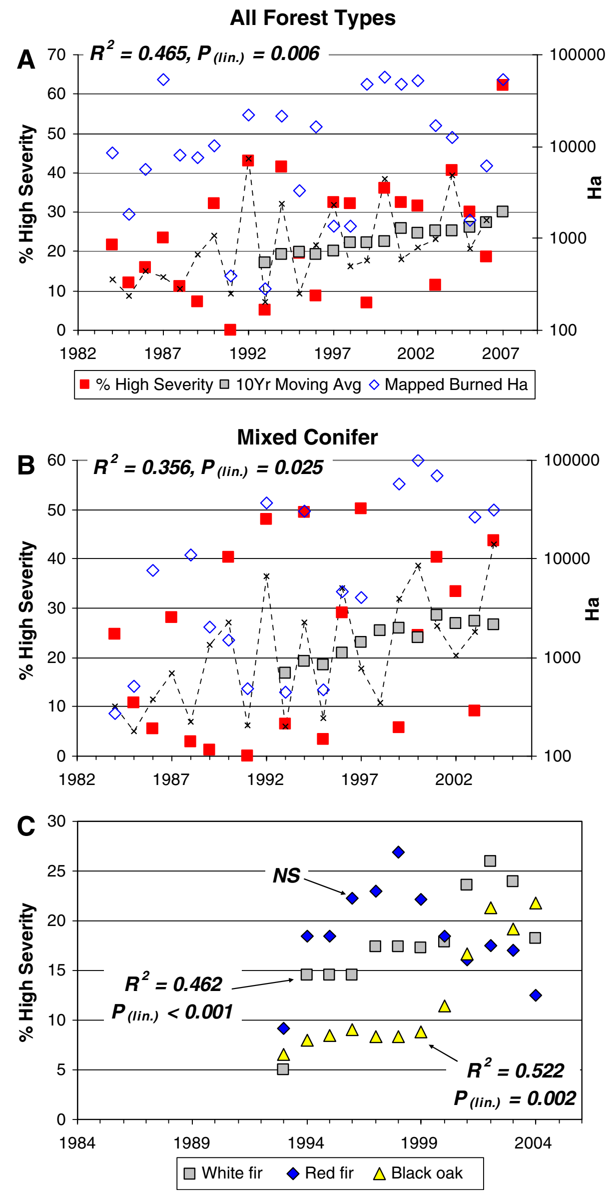

Figure 2. **(A)** Temporal trend in % area burned at high severity for all forest types combined (excepting pinyon-juniper) in the study region, 1984–2007, with the best-fit regression function (dashed line, 1984–2006 only, see text), 10-year moving average for % high severity, and burned area mapped (right-hand Y-axis). The first moving average data point represents 1984–1993. Pictured *P* values refer to the linear trend. The data were best fit by a 1st order autoregressive function, $P_{(AR1)} = 0.002$. The 1984–2006 model predicts 37% high severity in 2007, the actual (preliminary) value based on less than 1-year postfire imagery is 62%. **(B)** Temporal trend for mixed conifer forests, best fit by a 1st order autoregressive function, $P_{(AR1)} = 0.017$. **(C)** 10-year moving averages for % high severity for forests dominated by white fir, red fir, and black oak. The white fir data were best fit by a 4th order autoregressive function ($P_{(AR4)} = 0.001$), the black oak data by a linear function; red fir forests showed no significant trend across the study period. Data in (**B**) and (**C**) from USFS lands only.

period of analysis, a 10-year average of about 17% of the area affected in forest wildfires in the study region burned at high severity (that is, resulted in forest stand replacement); 23 years later the high severity component was approaching 30% of wildfire area (10-year mean; Figure 2A); restricting the analysis to USFS lands alone resulted in a similar pattern with a slightly weaker trend ($R^2 = 0.347$, $P = 0.004$). Although we will not have 1-year post-fire images for the 2007 fires until summer 2008, our preliminary analysis suggests that 2007 was the most severe fire year in the study region since the advent of Landsat imagery (Figure 2A). Different forest types showed different patterns within this general trend, with the proportion of high severity fire increasing at an average-or-above rate for most low- and middle-elevation forest types (for example, mixed conifer, white fir [*Abies concolor*], black oak [*Quercus kelloggii*]; Figure 2B, C), but at a below-average rate or not at all for high elevation forest types (for example, red fir [*A. magnifica*]; Figure 2C).

Patterns of Forest Fire Severity

During the 1984–2004 period, wildfires in lower- to middle-elevation conifer forests (for example, yellow pine [*P. ponderosa*, *P. jeffreyi*], and white fir) had highly severe effects on the forest canopy (that is, led to stand replacement) across 25–40% of the total burned area; higher elevation forests (for example, red fir) burned at lower severity (Figure 3, Table 1). Summing the severity data across years and forest types, high severity effects were measured on 26% of the burned acres analyzed, although there were marked differences between forest types (Table 1; note that the area analyzed for some of these types [for example subalpine, riparian, lodgepole pine] is too low to come to robust conclusions about severity patterns). Fires in the north study region (where temperatures are cooler, precipitation greater, and the fir component of the forest higher) tended to burn at somewhat higher severity than fires in the south region (Figure 3). Overall, our results corroborate general patterns reported from across the western US, where contemporary departures in fire activity and fire severity from pre-Euroamerican conditions range from high in many lower- to middle-elevation pine-dominated forests (that is, forests historically characterized by high frequency, low and mixed severity fire regimes), to low in wetter and/

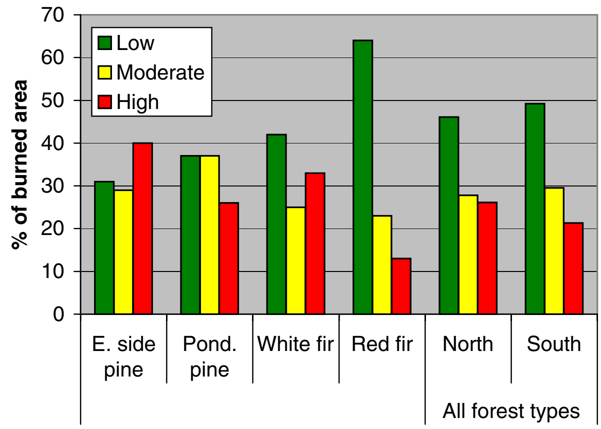

Figure 3. Low, moderate, and high severity fire as % of total burned area for four major forest types in the Sierra Nevada, 1984–2004, plus composites (excluding pinyon-juniper) for the north and south study regions. Only fires larger than 400 ha included (due to uneven spatial coverage of smaller fires); data from USFS lands only. E. side pine = *P. jeffreyi* and *P. ponderosa* stands, primarily on the east side of the Pacific Crest; P. pine = *P. ponderosa*, primarily on the west side of the Crest.

or higher elevation forest types (that is, forests historically characterized by lower frequency/higher severity fire regimes) (Sudworth 1900; Leiberg 1902; Agee 1993; Skinner and Chang 1996; Schoennagel and others 2004; Arno and Fiedler 2005; Noss and others 2006; Sugihara and others 2006).

High Severity Patch Size

Average and maximum sizes of contiguous areas (''patches'') of stand-replacing fire within conifer forest fires approximately doubled across the period of analysis, rising from about 2.8 ha and 50 ha (mean and mean maximum) in the first 10-year period to 5.3 ha and 118 ha in the last period ($R^2 = 0.325$ and 0.357; $P \leq 0.01$). High severity patch size is positively related to fire size ($R = 0.624$, $P < 0.001$), which likely has roots in the close correlation between large fire incidence and extreme fire weather (that is, hot, dry, windy conditions) (Agee 1993; Sugihara and others 2006).

Temporal Trends in Fire Size and Annual Burned Area, 1908–2006

The area burned annually in forest fires dropped from the 1930s through the middle of the century, but area burned has been increasing at a rapid rate since the 1970s and especially since the early 1980s (Figure 4A). Mean and maximum fire size have both risen since the beginning of the fire record, but the increase has entirely occurred over the last two or three decades—current 11-year averages for mean and maximum fire size are 40–70% higher than any values before 1980 (Figure 4B and C). Between 1984 and 2004, annual burned area increased in moister and higher elevation forests dominated by firs (moist mixed conifer, white fir, red fir; $R^2 = 0.276$–0.605), but there was no sig-

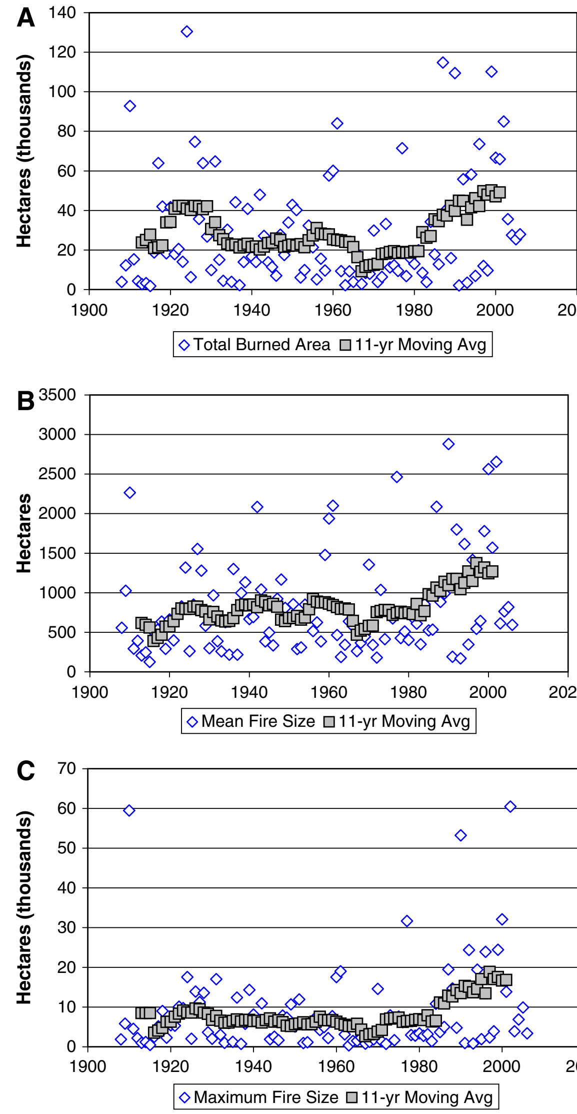

Figure 4. Temporal trends 1908–2006 in **(A)** annual total burned area, **(B)** average fire size, and **(C)** maximum fire size within the study region. Only fires larger than 40 ha included. Annual burned area is now at or above levels last seen before the advent of fire suppression policies; the 11-year moving averages for average and maximum fire sizes are now much higher than at any other time in the 99-year fire record.

nificant trend in most drier (that is, yellow pine-associated) lower- and middle-elevation forests.

Fire Relationships with Climate

Between 1908 and 2006, the Pacific Decadal Oscillation was positively but weakly correlated with mean and maximum fire size, with the relationship strongest at time lags of 5–7 years (fire area lagging behind the PDO; strongest relationship with 7-year lag: $R = 0.213$ and 0.227, respectively; $P < 0.05$). The Niño3.4 Index (ENSO) was also weakly correlated with mean and maximum fire size, but only under an 8-year lag ($R = 0.249$ and 0.269; $P < 0.05$). A +/− 5-year lag between PDO and fire occurrence was noted by Hessl and others (2004) for the Pacific Northwest, and their results and ours may represent a gradual-response feedback mechanism related to regional fuel moistures (or simply error in the PDO phase estimation). We are not certain how to interpret the lag in ENSO influence: given the positive correlation it may represent ENSO cyclicity and/or an artifact due to interaction with PDO; it may also simply be erroneous. The relationships between fire size and PDO and ENSO are not stable, and are characterized by temporal cycling in the strength and sign of the correlations (Figure 5). The strongest correlations between PDO and mean fire size occur as the PDO index approaches zero during a major phase shift from positive to negative or vice versa. The phase shift in 1947 was characterized by strong positive correlations between PDO and mean fire size, the 1923 shift by negative correlations. The ENSO-fire size correlations cycled in opposite phase to the PDO correlations from 1923 to 1947, but since then have more or less mirrored the PDO correlations (Figure 5). The strong increase in fire size beginning in the early 1980s has occurred as the strengths of both PDO and ENSO teleconnections have waned (Figure 5).

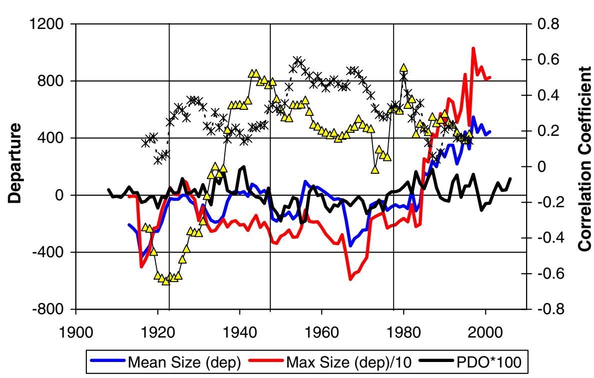

Figure 5. Temporal trends in mean and maximum fire size in the study region, 1908–2006 (11-year moving averages, shown as departures from the 99-year mean), compared to the Pacific Decadal Oscillation. Maximum fire size divided by 10, PDO multiplied by 100. Line with yellow triangles represents the strength of Pearson correlations between PDO (7-year lag) and log mean fire size, based on running 20-year periods 1908–2006; the dashed line with ''x's'' portrays the strength of correlations between ENSO (NIÑO 3.4 Index, 8-year lag) and log mean fire size, also based on running 20-year periods. Vertical lines = major phase shifts in the PDO. Correlations greater than 0.442 or less than −0.442 are significant at $P < 0.05$.

Between 1908 and 2006 there were significant increases in mean annual precipitation (+25.6 cm, $R^2 = 0.068$, $P = 0.009$) and mean minimum temperature (all four seasons, led by June–August [+1.6° C, $R^2 = 0.299$, $P < 0.001$]); there were no temporal trends in mean maximum temperatures. In the 1984–2004 period, fire severity in yellow pine-dominated forests (ponderosa and Jeffrey pine, dry mixed conifer) was best explained by springtime temperature minima, whereas severity in moister and higher elevation forest types dominated primarily by fir species (moist mixed conifer, true firs, subalpine) was best explained by spring and summer precipitation (Table 5). Correlations between fire severity and climate were strongest for forest types with a fir component (that is, in generally moister and higher elevation forests). Between 1908 and 2006, the annual number of recorded fires, fire size, and annual burned area in the study region were all positively related to spring and summer temperatures and, in the case of fire number and annual burned area, negatively related to winter precipitation. However, splitting the record into early (1908–1956), late (1957–2006), and very late (1982–2006) periods show that a shift in climate correlations has occurred. For the fire size variables, the early period was characterized by a positive correlation with spring temperatures, with the correlation shifting to summer temperatures in the late period (Table 5); annual burned area was negatively correlated with winter precipitation in the first half of the record, but with spring to autumn precipitation in the second half. For all four fire variables, the strength (adj. R^2) of the climate–fire relationship increases considerably from the first to the second half of the record and, for the fire size variables, the relationship is even stronger when only the last 25 years are considered (Table 5). Across the 99-year record, the proportion of variance in the fire variables explained by climate has more than doubled, from 9 to 23% in the early record to 38–49% in the late record. Fire size and

Table 5. Results of Stepwise Linear Regressions of Percent High Severity by Vegetation Type (Arcsin-Square Root Transformed; 1984–2004) and Fire Size Variables (Log-Transformed; 1908–2006) Versus Climate Variables

Dependent variable[a]	Period	Winter precip	Spring precip	Summer precip	Spring-fall precip	Spring max temp[b]	Spring min temp	Summer max temp	Summer min temp	p	R^2 (adj.)
Dry mixed conifer	1984–2004						0.715			0.000	0.512
Ponderosa pine	1984–2004						0.495			0.023	0.245
Jeffrey pine	1984–2004						0.380[c]			0.089	0.144
Moist mixed conifer	1984–2004		−0.732							0.000	0.537
Red fir	1984–2004		−0.539							0.011	0.290
Subalpine	1984–2004		−0.537							0.012	0.287
Mean fire size	1908–1956					0.303				0.034	0.092(0.073)
	1957–2006		−0.339						0.399	0.000	0.309(0.278)
	1982–2006		−0.375	−0.383						0.006	0.375(0.319)
Max fire size	1908–1956					0.392				0.005	0.153(0.135)
	1957–2006				−0.324				0.406	0.000	0.326(0.297)
	1982–2006		−0.397	−0.365						0.005	0.381(0.325)
Annual burned area	1908–1956	−0.259				0.371				0.004	0.211(0.177)
	1957–2006				−0.380				0.500	0.000	0.475(0.453)
	1982–2006				−0.435			0.401		0.001	0.487(0.441)

Values in climate columns are standardized regression coefficients. Except for[c], all predictor parameter estimates are significant at $P < 0.05$. [a]Dependent variable for forest types is % area burning at high severity. [b]''Max'' and ''min'' = mean maximum and mean minimum.

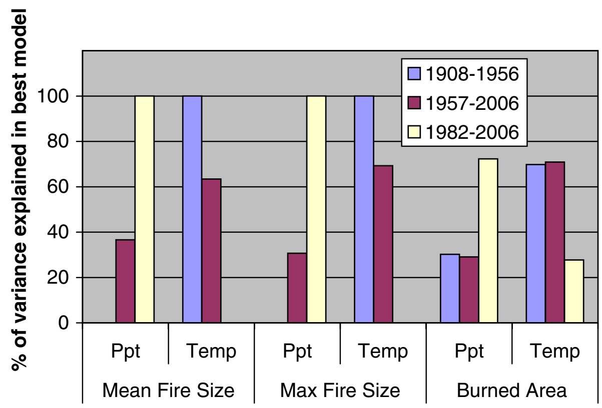

Figure 6. Comparison of proportions of variance explained by precipitation and temperature variables in the best-fit climate versus fire models reported in Table 5. Temperature accounts for all or most of the explained variance in the early period models; precipitation accounts for all or most of the explained variance in the latest period models.

burned area are increasing in concert with rising temperatures and precipitation, but whereas variance in fire size and annual burned area was primarily explained by temperature at the beginning of the record, it is now primarily explained by precipitation (Figure 6).

Discussion and Conclusions

Our results provide the first broad-scale, quantitative demonstration that the extent of forest stand-replacing fire is increasing across a significant part of the western US. Various lines of evidence suggest that contemporary fires in many low- and middle-elevation forest types in the study region burn at generally higher severities than before Euroamerican settlement (for example, Sudworth 1900; Leiberg 1902; Kilgore and Taylor 1979; Agee 1993; McKelvey and others 1996; Skinner and Chang 1996; Graham and others 2004; Arno and Fiedler 2005; Sugihara and others 2006; Beaty and Taylor 2007), and our data demonstrate that the magnitude of that departure is increasing with time. In our study area, forest types most affected by increasing fire severity are those which (1) form the majority of the National Forest landbase; (2) support most remaining habitat for a suite of old-forest obligate carnivores and raptors whose declining populations led to the SNFPA in the first place [for example, California spotted owl (*Strix occidentalis occ.*), goshawk (*Accipiter gentilis*), and fisher (*Martes pennanti*)]; (3) see the heaviest resource extraction and recreation use; and (4) are experiencing rapid growth in human population. Through their growing tendency to kill larger patches of canopy trees, contemporary fires are contributing to increasing levels of forest fragmentation. With continuing increases in the extent of high severity fire and high severity patch size, post-fire erosion, stream sedimentation, nutrient cycling, carbon sequestration and natural forest regeneration processes will also be increasingly impacted (Pickett and White 1985; Hobbs and others 1992; Gresswell 1999; Breashears and Allen 2002; Sugihara and others 2006; Allen 2007), and human safety is also a rising concern (Torn and others 1999; USDA 2004; CDF 2007b; Daniel and others 2007). The magnitude of these effects is still buffered by the fact that relatively few fires escape initial control (Calkin and others 2005), but the number, severity, and size of escaped fires is increasingly rapidly across the study region, even with tens to hundreds of million dollars spent annually on fire suppression.

Using a 1970–2003 dataset, Westerling and others (2006) showed a dramatic increase in large wildfire frequency in the western US beginning in the mid-1980s, centered in the northern Rockies and northern California (our study area plus adjoining coastal forests). Our data, which are from a finer spatial scale but much broader temporal scale and which include medium-sized fires, corroborate Westerling and others' (2006) findings and show that the post-1980 increase in wildfire activity is not restricted to fire frequency, but extends to fire size and annual burned area as well, at least in the study region. Although our fire severity dataset only begins in 1984, we hypothesize that the increases we see in the extent of forest stand-replacing fire in the study region are linked to these longer-term patterns. Like Westerling and others (2006) we find a significant relationship between climate and forest fire activity, but the temporal extent of our fire perimeter dataset allows us to discern three important trends in the nature of this relationship over time. First, early in the 20th century fire size and annual burned area in the study region responded largely to winter precipitation and springtime temperature (which influences snowmelt), but these fire variables now respond more directly to precipitation and temperature during the fire season itself. Second, fire number, size and annual burned area—and, at least over the last quarter-century, fire severity—have been rising in the study region even as regional precipitation has increased. Third, we document a strong increase in the relative importance of precipitation versus temperature in driving fire size and annual burned area over the last century. Heat, oxygen, and fuel are the fundamental extrinsic factors regulating fire combustion and maintenance (Sugihara and others 2006). Precipitation's direct influence on fire is negative, through the wetting of fuel, but there is also an indirect positive effect via increased fuels resulting from augmented vegetation growth. The temporal patterns we see in the climate–fire relationship are clearly due in part to increasing temperatures, but our results suggest a prominent role for increasing levels of forest fuels, presumably from a combination of fire suppression and precipitation effects. With snow water equivalents dropping, dates of peak snowmelt coming earlier, fire season lengthening, the summer drought deepening, and forest fuels at possibly all-time highs (Field and others 1999; Lenihan and others 2003; Miller and others 2003; Mote and others 2005; Westerling and others 2006), it may simply be that most low- and middle-elevation forestlands in the study region are ready and primed to burn, as long as the ignition coincides with a period of low fuel moisture.

Two studies using similar methodologies to our own have recently demonstrated increased severities of fire in WFU-managed areas in Yosemite National Park and the Gila Wilderness in New Mexico. The Yosemite study—which is nested within our larger study region—measured a large increase in fire severity in the mid-1980s, but little subsequent rise (Collins 2007). The probability of a fire reburning a previously burned area was limited by the time since last fire, suggesting that natural fire processes in the studied watershed were limiting fuels. This is very different from most of the study region as a whole, where long-term fire suppression has generated a fuels-rich environment. In the Gila study, Holden and others (2007) also saw a significant rise in fire severity in the period 1984–2004, but could not rule out precipitation effects in their analysis. Although our severity dataset also includes WFU fires (5% of sampled fire area, approximately equal to the proportion of WFU-managed area across all ownerships in the study region), it is composed primarily of suppressed wildland fires that escaped control, which is the nature of almost all forest fires in the western US today. In that respect, and due to the much larger spatial scale of our study, we believe the direction and strength of the patterns we see are probably more indicative of the current situation across much of the western US.

The Sierra Nevada and southern Cascades are geographically transitional between the southwestern and northwestern US, two areas where PDO and ENSO interactions with climate and fire are known to produce relatively predictable and opposite outcomes (Gershunov and Barnett 1998; Westerling and Swetnam 2003). Fire activity and size in the study area have a general tendency to increase when the PDO index is positive and ENSO is negative (''El Niño'')—joint conditions that usually lead to relatively warm and dry conditions in the Pacific Northwest—and to decrease during periods of negative PDO values and positive (''La Niña'') ENSO values, when the Northwest is typically cool and wet (Mantua and Hare 2002). Although the general tendency may be for the study region to follow Pacific Northwest patterns of climate–fire teleconnections, the transitional nature of the area leads to a schizophrenic association with these phenomena, such that some decades are characterized by northwestern type PDO × ENSO effects, and others by southwestern type PDO × ENSO effects (Westerling and Swetnam 2003). This decadal-scale variability is reflected in the shifting sign and strength of correlations between fire size, PDO and ENSO in Figure 3. It is noteworthy that the pronounced increases in fire size that commenced in the study area in the early 1980s began under conditions of relatively strong PDO and ENSO teleconnections, but have continued as the influence of these teleconnections has moderated considerably.

Two key questions arise from this and other studies related to recent wildfires in the western US. First, what factors are driving the widespread increases we are seeing in fire activity and severity? Second, what role can, or should, land management play in stemming or modifying these trends? Fuels, climate, and fire management policy all play major roles in determining the outcomes of fire in western forests, and all of these factors are interlinked (Agee 1993; McKelvey and others 1996; Schoennagel and others 2004; DeWilde and Chapin 2006; Sugihara and others 2006; Falk and others 2007). Fire cannot burn without fuel, and the natural fire–fuel relationship is therefore inherently self-limiting, at least at some spatiotemporal scale. Although regional climate explains much of the variance in the fire variables we measured, the nature of the fire–climate relationship suggests that the increases we measured in forest fire size, burned area and severity may be largely driven by increases in forest fuels, due to human fire suppression and possibly to increased precipitation as well. For nearly a century, national fire management policies have mandated immediate suppression of almost all wildland fires, with great success: across California, current annual burning affects only 6% of the area burned annually before Euroamerican settlement (Stephens and others 2007). The unintended consequences of this strategy—in conjunction with certain forest management practices, such as the discredited custom of leaving large amounts of timber slash onsite after logging—have been to dramatically increase forest fuels in yellow pine and mixed conifer forests across the West, and to predispose to high severity those fires which do escape control (see DeWilde and Chapin [2006] for similar results from Alaska). Our results and the results of others (Field and others 1999; Miller and Urban 1999; Lenihan and others 2003; Brown and others 2004; Calkin and others 2005; Running 2006; Westerling and others 2006; Morgan and others 2008) show that the importance of climate in regulating wildfire is growing across the western United States. With respect to fire severity and size in our study region—where most of the forested landscape historically supported relatively high frequencies of low to moderate severity fire and thus fairly low fuel loadings—we hypothesize that this pattern is to a large extent an effect of the current and continuing absence of an agent to remove forest fuels at a rate compatible with their accumulation. In light of recent alarming projections for increased temperatures and fire-season length by the end of the century (Field and others 1999; Lenihan and others 2003; Calkin and others 2005; Westerling and others 2006; IPCC 2007), a major rethinking of current fire and fuels management strategies may be in order.

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