

CHAPTER 13

CLIMATE VARIABILITY, CLIMATE CHANGE, AND WESTERN WILDFIRE WITH IMPLICATIONS FOR THE URBAN–WILDLAND INTERFACE

William S. Keeton, Philip W. Mote and
Jerry F. Franklin

ABSTRACT

Climate change during the next century is likely to significantly influence forest ecosystems in the western United States, including indirect effects on forest and shrubland fire regimes. Further exacerbation of fire hazards by the warmer, drier summers projected for much of the western U.S. by climate models would compound already elevated fire risks caused by 20th century fire suppression. This has potentially grave consequences for the urban–wildland interface in drier regions, where residential expansion increasingly places people and property in the midst of fire-prone vegetation. Understanding linkages between climate variability and change,

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therefore, are central to our ability to forecast future risks and adapt land management, allocation of fire management resources, and suburban planning accordingly. To establish these linkages we review previous research and draw inferences from our own retrospective work focused on 20th century climate–fire relationships in the U.S. Pacific Northwest (PNW). We investigated relationships between the two dominant modes of climate variability affecting the PNW, which are Pacific Decadal Oscillation (PDO) and El Niño/Southern Oscillation (ENSO), and historic fire activity at multiple spatial scales. We used historic fire data spanning most of the 20th century for USDA Forest Service Region 6, individual states (Idaho, Oregon, and Washington), and 20 national forests representative of the region's physiographic diversity. Forest fires showed significant correlations with warm/dry phases of PDO at regional and state scales; relationships were variable at the scale of individual national forests. Warm/dry phases of PDO were especially influential in terms of the occurrence of very large fire events throughout the PNW. No direct statistical relationships were found between ENSO and forest fires at regional scales, although relationships may exist at smaller spatial scales. However, both ENSO and PDO were correlated with summer drought, as estimated by the Palmer Drought Severity Index (PDSI), and PDSI was correlated with fire activity at all scales. Even moderate ($\pm 0.3^{\circ}\text{C}$ decadal mean) fluctuations in PNW climate over the 20th century have influenced wildfire activity based on our analysis. Similar trends have been reported for other regions of the western U.S. Thus, forest fire activity has been sensitive to past climate variability, even in the face of altered dynamics due to fire suppression, as in the case of our analysis. It is likely that fire activity will increase in response to future temperature increases, at the same or greater magnitude as experienced during past climate variability. If extreme drought conditions become more prevalent we can expect a greater frequency of large, high-intensity forest fires. Increased vulnerability to forest fires may worsen the current fire management problem in the urban–wildland interface. Adaptation of fire management and restoration planning will be essential to address fire hazards in areas of intermingled exurban development and fire-prone vegetation. We recommend: (1) landscape-level strategic planning of fire restoration and containment projects; (2) better use of climatic forecasts, including PDO and ENSO related predictions; and (3) community-based efforts to limit further residential expansion into fire-prone forested and shrubland areas.

INTRODUCTION

Wildfire dynamics in portions of the western United States have been dramatically altered from pre-European settlement conditions. Increased fire hazards due to 20th century fire suppression and other human activities have serious implications for the urban–wildland interface, including risks to human safety and property. These trends necessitate difficult resource management and planning decisions as communities and housing expand into previously undeveloped, fire-prone areas (Cova, Sutton, & Theobald, 2004; GAO, 1999). Superimposed on these trends are the potential effects of climate change, which are predicted to increase the frequency and severity of drought conditions (Brown, Hall, & Westerling, 2004) and extreme fire weather (McKenzie, Gedalof, Peterson, & Mote, 2004) across the U.S. southwest, interior Great Basin, and northern Rocky Mountain region in particular (Brown et al., 2004). Climate-related fire risks have the potential to compound the present fire management problem along the urban–wildland interface. Understanding linkages between climate variability and change, therefore, are central to our ability to forecast future risks and adapt land management, allocation of fire management resources, and suburban planning accordingly. To establish these linkages we review previous research and draw inferences from our own work focused on 20th century climate–fire relationships in the U.S. Pacific Northwest (PNW, Fig. 1).

In the western United States interactions between climate variability and fire are likely to be important drivers of forest ecosystem responses to climate change. For this chapter we define climate variability as fluctuations in climatic conditions over multiple time scales and primarily attributed to natural processes. Climate change is treated as future changes in the global climate system, primarily related to anthropogenic causes (IPCC, 2001). Our work has focused on large-scale modes of climatic variation over the tropical and north Pacific Ocean, including El Niño/Southern Oscillation (McPhaden et al., 1998) and the Pacific Decadal Oscillation (PDO) (Mantua, Hare, Zhang, Wallace, & Francis, 1997), respectively. We have used a retrospective approach to understand how fire frequency and intensity responded to past climatic fluctuations. This improves our ability to predict how disturbances, and related fire risks along the urban–wildland interface, will respond to future climate changes, especially alterations of climate-related stressors, like extreme drought events, for which we can find historic analogues.

Relationships between the PDO and fire activity in the interior Northwest have been identified by previous studies that relied on dendrochronological



Fig. 1. The Climate Impacts Group focuses on the Columbia River Basin (Out-lined) and the States of Washington, Oregon, and Idaho. Figure Courtesy of Robert A. Norheim, Climate Impacts Group, University of Washington, Seattle, WA.

(i.e. tree ring) methods to establish fire and climate histories extending back to 1700 AD (Gedalof & Smith, 2001; Hessl, McKenzie, & Schellhaas, 2004). Other recent work has used climate projections to predict possible 21st century changes in the timing, duration, and intensity of climate variables related to western U.S. forest fire danger (Brown et al., 2004; McKenzie et al., 2004). Both approaches have found clear linkages between fire risks, past climate variability, and future climate change, with extended drought during the fire season acting as the fundamental climate mechanism associated with elevated fire hazards. Our retrospective research attempts to support these findings and predictions using historical, documented records of 20th century fire activity and direct measurements of climate variability. This work was undertaken as part of the regional assessment of climate variability and climate change impacts on the PNW (Mote et al., 2003), part of the National Assessment program (NAST, 2000). Selected elements and

summations of this work, undertaken by the interdisciplinary Climate Impacts Group at the University of Washington, have previously been reported elsewhere (Keeton, Franklin, & Mote, In press; Mote et al., 1999a; Mote, Keeton, & Franklin, 1999b; Mote et al., 2003; Parson et al., 2001), but this paper is the first to report our findings in full.

Climate change is predicted to have direct and indirect effects on forest ecosystems (Fig. 2). Direct effects include altered physiological processes due to changes in temperature and precipitation regimes as well as CO₂ enrichment. These are predicted to cause changes in the distribution, composition, and productivity of forest ecosystems nationwide (NAST, 2000). Potential indirect effects include altered natural disturbance regimes, including changes in the frequency, intensity, and spatial extent of fire, insect, disease, and wind disturbances (Keeton et al., In press).

Over the near-term, climate-driven natural disturbances may be even more important than the direct effects of climate change in causing abrupt or rapid forest ecosystem responses (Fosberg, Mearns, & Price, 1992; Overpeck, Rind, & Goldberg, 1990; Ryan, 1991). Changes in vegetation composition and structure may be especially rapid on sensitive sites or near the limits of species' ranges (Allen & Breshears, 1998; Brubaker, 1988).

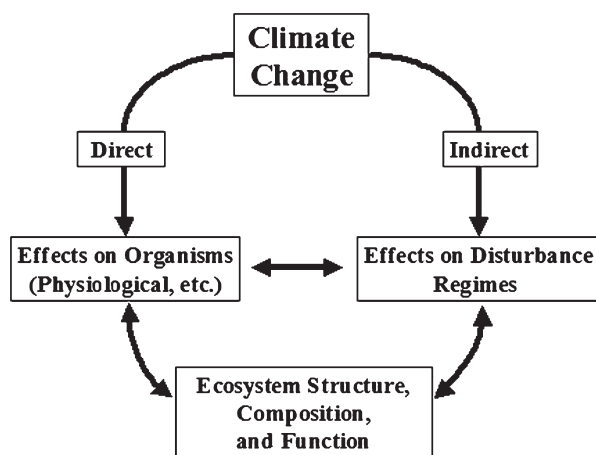


Fig. 2. Climate Change is Predicted to Impact Forested Ecosystems both through Direct Effects on Organisms and Indirect Effects on Natural Disturbance Regimes (e.g., Fire, Insects, Pathogens, and Wind). Feedback relationships among these pathways of change contribute collectively to increased fire risks in the urban-wildland interface. Modified from Franklin et al. (1991).

Established forests often can resist climatic variability both because they ameliorate microclimatic conditions within the forested ecosystem and because mature trees can survive extended periods of less favorable climate (Brubaker, 1986; Dale & Franklin, 1989; Franklin et al., 1991). High-intensity disturbances, however, have the potential to reset stand development to the establishment stage (Franklin et al., 2002), which is the stage most sensitive to adverse environmental conditions, such as drought and heat (Brubaker, 1986). Stand-replacing disturbances are likely to cause more rapid transitions in ecosystem composition and structure over the near-term than are the direct changes in tree growth rates alone (Franklin et al., 1991; Overpeck et al., 1990). It is, therefore, critical to understand relationships between disturbance dynamics and climate variability if we are to accurately predict both rates and pathways of future ecosystem change as well as associated fire risks.

While mean climate varies considerably across the Northwest, interannual variations in climate are strongly correlated within the region (Mote et al., 2003). Warm versus cool years tend to be experienced similarly throughout the region. This regional coherence permits us to focus on temporal fluctuations in the regional average anomalies. Year-to-year global climatic variations are dominated by El Niño/Southern Oscillation (ENSO), an irregular oscillation of the tropical atmosphere and ocean with a period of 2 to 7 years (McPhaden et al., 1998). Interannual variations in forest fire activity in the U.S. Southwest are significantly correlated with the ENSO phenomenon (Swetnam & Betancourt, 1990). In the Northwest, the influence of ENSO on regional climate is rivaled by another such irregular variation, this one in the north Pacific basin: the Pacific Decadal Oscillation (PDO). By calculating empirical orthogonal functions (EOFs) of monthly Pacific sea surface temperature (SST) north of 20° N, Mantua et al. (1997) identified the PDO as the dominant mode of variability on interannual timescales in the north Pacific (Fig. 3).

The PDO is a pattern of Pacific SST anomalies whose positive phase is associated with cold anomalies in the central Pacific and warm anomalies along the west coast of North America. It resembles the SST pattern that usually coincides with ENSO, but has different temporal characteristics. A time series of the loading of the first EOF (Fig. 3) exhibits slow variations in which the dominant sign remains the same for 20–30 years. It was in the negative phase from about 1900 (when a few reliable SST measurements began to be available) to 1925 and from 1945 to 1977, and in the positive phase from 1925 to 1945 and from 1977 to 1999. Since 1999 PDO has returned to its negative phase. Warm phases of ENSO and PDO coincide

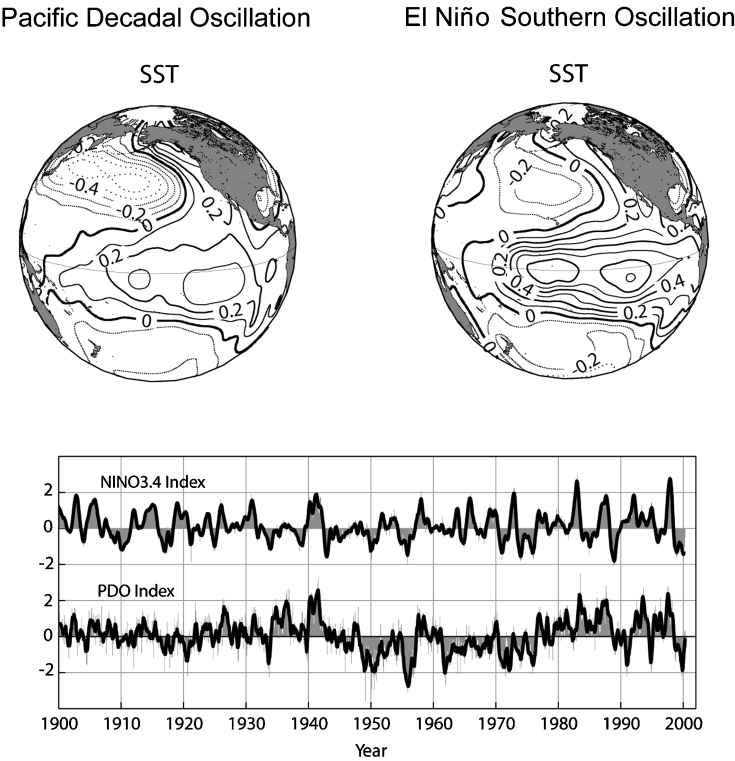


Fig. 3. Spatial Pattern of Anomalies in Sea Surface Temperature (SST; degrees Celsius) Associated with the Warm Phase of PDO (Left) and ENSO (Right). Note that the main center of action for the PDO is in the north Pacific, while the main center of action for ENSO is in the equatorial Pacific. Time histories of the PDO and ENSO patterns are shown below. When the Nino 3.4 or PDO index is positive, the SST anomalies resemble those shown in the contour plots. When the index is negative, the SST anomalies would be reversed. Images provided by the University of Washington, Climate Impacts Group.

with winter and spring weather that is warmer and drier than average in the PNW, and cool phases coincide with cooler, wetter weather.

We investigated relationships between the two climate time series, ENSO and PDO, and 20th century fire activity in the PNW at multiple spatial scales. Our hypothesis was that relationships are scale dependent due to spatial variation in mechanistic relationships linking climate and fire. These two climate patterns (ENSO and PDO) are useful for our purposes in at

least two ways. First, together they provide robust predictability in seasonal forecasts for the region. Second, the multi-decadal timescale of the PDO may provide a useful surrogate for anthropogenic climate change. For forest ecosystems the persistence of warmer-drier or cooler-wetter conditions over 20–30 years is likely to produce a higher magnitude response than do single, anomalous years (Mote et al., 2003).

METHODS

We analyzed relationships between 20th century forest fire activity and climatic variability at three spatial scales: regional (USDA Forest Service Region 6: Washington and Oregon), individual states (Idaho, Oregon, and Washington), and individual national forests within the PNW. These geopolitical scales were defined by the available historic fire datasets. We selected 20 national forests for analysis that are representative of the region's physiographic provinces and precipitation divisions.

We collected data on forest fires in Idaho, Washington, and Oregon and correlated the year-to-year variations with ENSO and PDO. The fire data include area burned, area monitored, and number of lightning vs. human-caused fires. Fire data time series were compiled from USDA Forest Service annual forest fire reports and data from the National Archives covering 1905–2000 for the region (Fig. 4), 1916–2000 for individual states, and 1922–2000 for individual national forests. They are considered to be independent data sets, because state- and regional-level data series were collected using different methods and, consequently, do not sum to increasingly coarser scales. For the state and regional data, we constructed Burn Area Indexes (BAI) by normalizing the area burned each year by the area monitored in that year, since this fluctuated over time. The indexes were calculated as follows:

$$\text{BAI} = \left(\frac{\text{hectares burned}}{\text{hectares monitored}} \right) \times 10,000$$

As a measure of ENSO we used the Nino3.4 index. For the PDO we used six-month means (October–March or “winter,” and April–September or “summer”) of the monthly time series generated by Mantua et al.'s (1997) EOF analysis and subsequent monitoring.

Our correlation analyses also included comparisons between climate and fire time series and the Palmer Drought Severity Index (PDSI), which is an estimate of accumulated soil moisture deficit (Palmer, 1965). We used PDSI

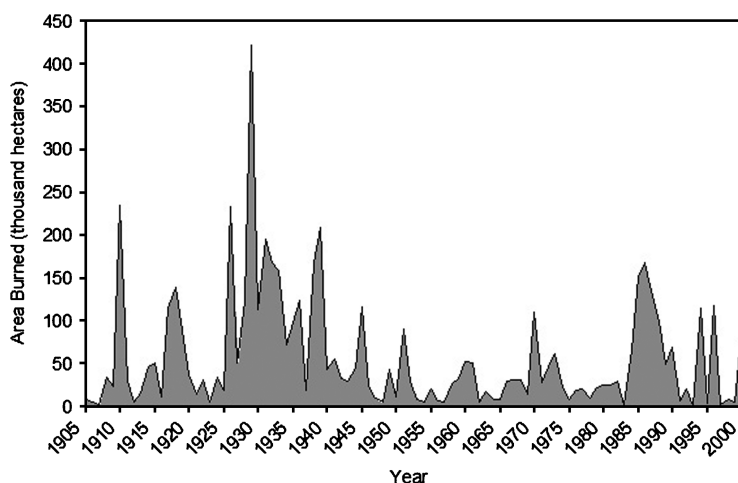


Fig. 4. Area Burned by Forest Fires in USDA Forest Service Region 6 (Washington and Oregon) from 1905 to 2000. The data shown have not been normalized to account for fluctuations in the area monitored over time.

as an indicator of drought conditions, our hypothesized intermediary mechanism. Linear regression analysis was used for statistical testing of fire data against time series for ENSO, PDO, and PDSI. Residuals were examined to confirm assumptions of normality using the Wilk–Shapiro test. Significance levels were estimated using Monte Carlo simulations. We used a 90% confidence level to determine significance due to the high degree of noise inherent in climate data. We used the Durban–Watzin test, performed on the residuals resulting from each regression combination, to identify cases where correction for autocorrelation was necessary. We confirmed these results using an autocorrelation test developed by Bretherton, Widmann, Dymnikov, Wallace, and Blade (1999), and corrected degrees of freedom and significance levels accordingly.

Scatter plots of BAI by year showed discrete thresholds of separation between years in which relatively small total areas were burned and years in which large areas were burned. We used these thresholds to define “large-fire” years at the scale of states ($> 80,000$ ha burned) and unit of the National Forest System (> 400 ha burned). We conducted additional analysis on the number of “large-fire” years at these scales that occurred during either warm/dry or cool/wet phases of ENSO or PDO, grouped as categorical data into observed vs. expected distributions. To test for differences

between distributions we used a log likelihood-ratio goodness of fit (*G*-test) with the Yates correction for continuity. This test approximates the X^2 statistic, but it is more robust than the Chi-square goodness of fit test when certain conditions are met (Zar, 1996), as was the case with our data.

RESULTS

Pacific Decadal Oscillation

Regional scale BAI is closely correlated with the PDO (Table 1). Forest fires were much more extensive in the USDA Forest Service Region 6 during the 1925–1945 warm phase of PDO than during the cool phases before and after that (Mote et al., 1999b). The resurgence of fire activity in the late 1980s was consistent with the warm–dry phase of the PDO, but also correlates with increased fire hazards due to fire suppression. When year-to-year values of the PDO are considered, however, we find a significant correlation with BAI. The winter PDO index has a correlation coefficient of 0.31, which is statistically significant at the 99% level using Monte Carlo methods. Summer PDO was not significantly correlated with BAI.

The increased tendency for forest fires in warm-phase PDO years holds at finer spatial scales (Table 1). Warm-phase PDO is positively correlated with BAI for Washington ($P = 0.003$). However, for Idaho ($P = 0.055$) and Oregon ($P = 0.062$) this relationship was only moderately strong, which signals possible interactions with other scale-dependent sources of variability. At the scale of individual states, correlations with PDO shift by state when we restrict our analysis to exceptional years in which very large areas

Table 1. Statistical Results Correlating Pacific Decadal Oscillation against Burn Area Index at Regional (1905–2000) and State (1916–2000) Scales. Results of Autocorrelation Analyses of Time Series are also Reported.

	USFS Region 6	Oregon	Washington	Idaho
Correlation coefficient (<i>R</i>)	0.31	0.211	0.42	0.21
Degrees of freedom	89	80	80	80
<i>P</i> value	0.003	0.062	<0.001	0.055
Durban Watzin (DW) statistic	1.622	1.533	1.688	2.245
DW critical value at 99% significance level	1.550	1.530	1.530	1.530
Autocorrelation probability (%)	<1	<1	<1	<1

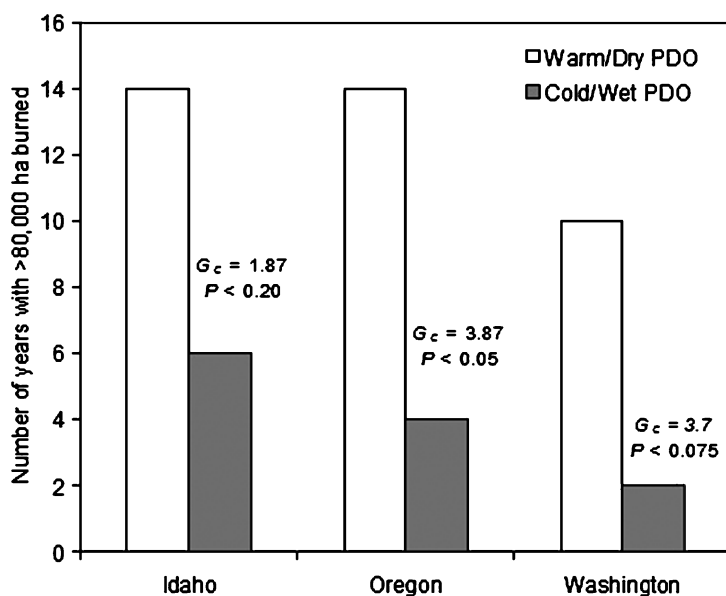


Fig. 5. The Number of Years with >80,000 Hectares Burned During with either Warm/Dry (Positive) Phases or Cool/Wet (Negative) Phases of Pacific Decadal Oscillation: 1916–2000. Statistical results are based on the log likelihood ratio goodness of fit test.

(e.g., >80,000 hectares) were burned (Fig. 5). The differences in numbers of large-fire years for warm-dry vs. cool-wet PDO are statistically significant ($\alpha = 0.10$) for Oregon ($P < 0.05$) and Washington ($P < 0.075$), although not for Idaho ($P < 0.2$). Annual area burned at the scale of individual national forests is also more prevalent during the warm phases of PDO, and was correlated ($\alpha = 0.05$) with the values of the PDO for 8 of the 20 national forests selected (Table 2). All of the national forests positively correlated with warm-phase PDO were either in the semi-arid interior Northwest, the southern Cascade Range, or the Klamath-Siskiyou Mountains. When we restrict our analysis to extreme fire years, for instance those years in which greater than 400 hectares burned on an individual national forest, the warm phase of PDO generally increases the likelihood of large fires, although these relationships are only statistically significant at the 90% level and some forests have fewer large fires in the warm phase of PDO. The tendency for warm phase of PDO to increase the likelihood of large fires holds when the results are aggregated for all forests. Some forests do not show the same sensitivity

Table 2. Correlations Between Pacific Decadal Oscillation and Annual Burn Area at the Scale of Selected National Forests in Washington, Oregon, and Idaho. Note that the Number of Years of Available Data Varies by National Forest; All Time Series Run to the Year 2000.

	Correlation Coefficient	No. of Years in Data Set	Significant Relationship with PDO at 95% Level?
Colville	0.284	44	Yes
Deschutes	0.195	78	Yes
Fremont	0.236	72	Yes
Gifford Pinchot	0.186	49	No
Malheur	-0.035	51	No
Baker-Snoqualmie	0.12	51	No
Mt. Hood	0.108	53	No
Nez Perce	0.304	45	Yes
Ochoco	-0.068	70	No
Okanogan	0.181	56	No
Olympic	-0.132	58	No
Rogue River	0.178	47	No
Siskiyou	0.276	63	Yes
Siuslaw	0.133	52	No
Umatilla	0.234	63	Yes
Umpqua	0.24	57	Yes
Wallowa-Whitman	0.154	54	No
Wenatchee	0.209	70	Yes
Willamette	-0.127	69	No
Winema	0.143	34	No

to the PDO that the regional average shows; as with meteorological data (Mote et al., 2003), variability exists within the overall regional patterns.

The occurrence of drought, as measured by the PDSI, during warm phases of PDO may explain the linkage between PDO and wildfire (Fig. 6). PDSI is correlated with both PDO and wildfire activity. Regional PDSI and PDO values are moderately well correlated ($P < 0.10$), and PDSI values indicating drought conditions are correlated with the BAI for Idaho ($P < 0.10$), Oregon ($P < 0.01$), and Washington ($P < 0.01$).

El Niño/Southern Oscillation

At the regional scale, the 20th century forest fire data for the PNW show little relationship to ENSO. We found no statistically significant relationship

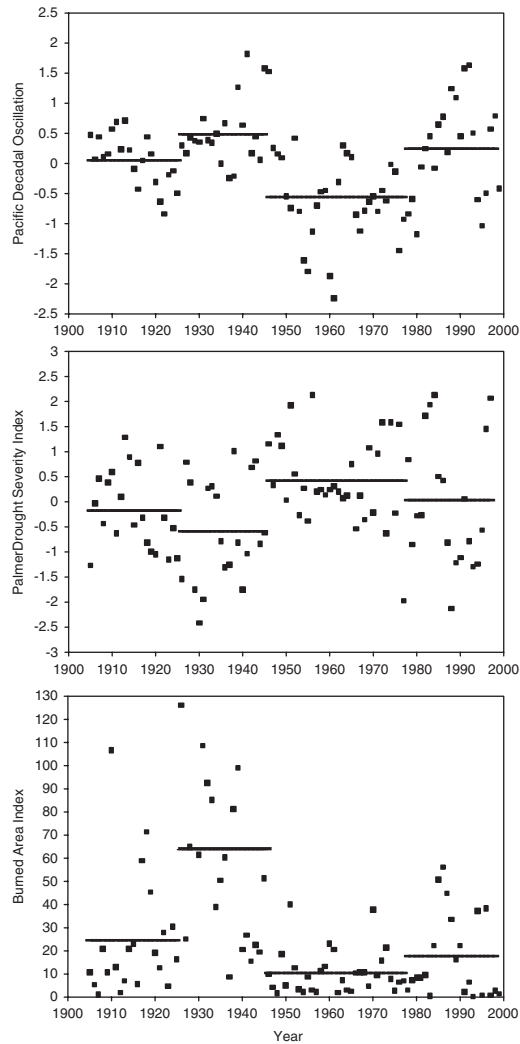


Fig. 6. Top: The Winter Pacific Decadal Oscillation (PDO) Index and Its Mean for each Phase Shift over the 20th Century. Positive values of the PDO are indicative of its warm/dry phase, whereas negative values are typical during cool/wet phases. Middle: The winter Palmer Drought Severity Index (PDSI) and its mean during each phase of the PDO. Negative values of the PDSI are indicative of drought conditions. Bottom: Burn Area Index for Washington and Oregon (USDA Forest Service Region 6) and its mean during phases of the PDO. Note the correspondence in means (lines) for each of the three indexes.

between the two. However, there may nevertheless be an indirect relationship through drought mechanisms. The PDSI is influenced by ENSO ($R = -0.48$, $P < 0.05$), and in turn the PDSI is a fairly good predictor of how extensive wildfires will be in a given year.

Relationships between ENSO and BAI were variable at the state level. ENSO was not correlated with BAI for Oregon and Washington; the variables were moderately well correlated for Idaho ($P < 0.10$). At the scale of national forests, statistically significant correlations between BAI and warm phases of ENSO were found for only 5 of the 20 national forests analyzed. These were the Deschutes ($R = 0.27$, $P < 0.01$), the Fremont ($R = 0.36$, $P < 0.001$), the Rogue, the Siskiyou ($R = 0.17$, $P < 0.1$), and the Winema ($R = 0.32$, $P < 0.01$).

DISCUSSION

Pacific Decadal Oscillation

The Pacific Decadal Oscillation (PDO) influences forest fire activity at multiple spatial scales in the PNW based on our analysis. These relationships are variable at within-region scales. Warm-dry phases of PDO are particularly influential in terms of the occurrence of very large fire events throughout the PNW. It is noteworthy that summer PDO was not correlated with BAI, probably because it has too little influence over summer climate in the PNW. Winter PDO, by comparison, has a large influence on snowpack and thus, indirectly, on summer soil moisture availability in montane systems.

All of the national forests positively correlated with warm-phase PDO were in portions of the PNW characterized by lower precipitation (compared to coastal forests) and dry coniferous forests at low to moderate elevations. Fire in these forest systems/sub-regions appears to show greater sensitivity to PDO. Linkages between fire occurrence and PDO have been reported for the interior Northwest by previous studies based on analysis of tree ring records (Hessl et al., 2004; Heyerdahl, Brubaker, & Agee, 2001). For instance, atmospheric circulation patterns from May to August appear to have influenced the area burned by wildfire in interior forests for several centuries prior to the advent of fire suppression (Gedalof, Peterson, & Mantua, 2005). Our results suggest that the influence of PDO on annual burn area and sub-regional variation in fire activity is evident in the historic record as well.

Before 20th century fire trends can be attributed to climate variability there are other confounding influences that warrant consideration. Some of the decline in the area burned by wildfires during the cool phase of PDO after 1945 and some of the increase in the most recent warm phase may be related to effects of fire suppression programs. Fire suppression retarded fire activity beginning mid-century, leading to elevated fire hazards in the later 20th century as fuel loads, tree densities, and in-growth of fire-prone species increased. Although these changes were prevalent primarily in dry, interior Northwest forests (Quigley, Haynes, & Graham, 1996) they may have created a confounding trend relative to PDO. However, that our results are nevertheless robust is suggested by historic precipitation and streamflow data (Mote et al., 1999a) which show that the 1925–1945 period was unusually dry in the Columbia River Basin. That period also had the largest amount of fire activity during the 20th century. Moreover, PDO and BAI are correlated when considering inter-annual oscillations in both indexes, suggesting a relationship that is apparent even when superimposed on inter-decadal trends.

El Niño/Southern Oscillation

Our analysis at the regional-scale showed no statistically significant correlations between 20th century area burned and ENSO, despite the connection between ENSO and PDSI, and variable relationships at smaller scales. Given the strong relationship between ENSO and burned area in the U.S. Southwest (Swetnam & Betancourt, 1990), and the strong statistical relationship between ENSO and climate in the Northwest, this result was unexpected, although previous studies have also found an “ambiguous relationship between ENSO and fire occurrence” in portions of the Northwest (Hessl et al., 2004). We suggest two possible explanations for our results. First, ENSO has a strong effect on the Southwest’s rainfall, which often leads to greater fuel accumulations during positive phases of ENSO. As a consequence, late-summer fires tend to increase during and after a cool wet El Niño winter. In contrast, high fuel loadings are always present in temperate forests west of the Cascade Range and in those forest types characterized by a multi-storied structure in the interior Northwest. Thus, sub-regional variations in climate–fire relations may be obscured when fire activity is averaged at the regional level. Second, 20th century fire suppression may have obscured climate–fire relationships on inter-annual time scales (Hessl et al., 2004; Westerling & Swetnam, 2003).

There is reason to suspect linkages between ENSO and wildfire at smaller geographic scales. Some watersheds in the Blue Mountains of northeastern Oregon and southeastern Washington show a statistical relationship between annual extent of low-severity fires and El Niño events; others do not (Heyerdahl et al., 2001; Heyerdahl, Brubaker, & Agee, 2002). Positive correlations were attributed to ENSO modulation of snowpack formation. It is interesting also to consider the ecological characteristics of the five national forests where we found burn area to be correlated with warm/dry phases of ENSO. All five forests are characterized by low to moderate severity fire regimes, with precipitation regimes among the driest and most drought-prone in the region. However, several national forests with similar characteristics did not show correlations between burn area and ENSO.

Tree-ring studies, using pre-fire suppression fire data, have proven more effective at exploring linkages between ENSO and forest fires. This research has found a strong polarity between the Southwest and Northwest United States in drought responses to ENSO (Westerling & Swetnam, 2003). El Niño (ENSO negative) events tend to be hot and dry in the Northwest vs. cool and wet in the Southwest; the inverse is true of La Niña (ENSO positive) events. It has also found indications of increased forest fires in the Northwest during El Niño years, especially if there is a sequence of multiple El Niño (hot/dry) years, during which fuels become successively more desiccated. There are interactions between ENSO and PDO: fire activity is highest during the coincidence of hot/dry phases of both climate patterns (Westerling & Swetnam, 2003).

Drought as an Intermediary Mechanism

Our results are consistent with other studies establishing drought as a mechanism linking climate variability, especially PDO, with fire activity in the Northwest (Hessl et al., 2004; McKenzie et al., 2004; Westerling & Swetnam, 2003). We found the PDSI to be an intermediary variable correlated with both 20th century climate variability and burn area. This is consistent with our understanding of drought-related disturbance dynamics. Reduced tree vigor during drought years makes trees more susceptible to both insect attack and wildfire damage (Swetnam & Lynch, 1993). Fire hazards are predicted using PDSI because the index is predictive of fuel conditions, such as fuel moisture content and flammability, in general (Westerling & Swetnam, 2003).

Drought appears to have had an important influence on fire occurrence for several hundred years in the interior Columbia Basin (Hessl et al., 2004) and U.S. Southwest (Westerling & Swetnam, 2003) prior to the advent of fire suppression. In dry, interior Northwest forests there may have been interactions between periods of higher moisture that increased fuel production, followed by drought years that created conditions necessary for fire ignition and spread. Major fire events in forests west of the Cascade Range also coincided with prolonged periods of drought during the last 1,000 years, based on fire history reconstructions (Hemstrom & Franklin, 1982). Thus, drought is, both statistically and biophysically, a primary mechanism linking climate variability and wildfire activity.

Synoptic-Scale Fire Weather

There are limitations in our ability to link climate variability with fire activity and predict future risks associated with climate change. Indicative of this uncertainty is that the historic fire data are “noisy;” substantial fire activity occurs across both cool-wet and warm-dry phases of PDO. This signals the critical importance of fire weather that may be unrelated to larger modes of climate variability, such as PDO. In particular, there are synoptic-scale (on the order of 2000 km) weather events that are strongly associated with fire outbreak and spread. These “fire weather” sequences occur even during otherwise wet years and weaken the connection between years with many large fires and seasonal-scale climate variations like those associated with ENSO and PDO.

A number of studies have described a synoptic-scale sequence of weather events leading to lightning-caused ignition and fire spread. This sequence of weather events has been described for boreal forests in Canada (Johnson & Wowchuk, 1993; Jones, 2000), coastal temperate coniferous forests in the PNW (Huff & Agee, 1980; Pickford, Fahnestock, & Ottmar, 1980), ponderosa pine forests in the Southwest (Swetnam & Betancourt, 1990), and for the entire United States by sub-region (Heilman, Eenigenberg, & Main, 1994). For the PNW, the weather sequence begins with the development of an atmospheric high-pressure upper-level ridge, also known as a blocking high-pressure system. The high-pressure system may last a month or more, during which time precipitation and humidity are low, temperatures are high, and winds are light. These conditions leave fuels dry and vegetation severely stressed. When the high-pressure system partially or fully breaks down, convective storms can lead to lightning-caused ignition which, when

combined with higher wind speeds, can lead to fire spread through the now flammable fuels (Lenihan, Daly, Bachelet, & Neilson, 1998). Other work also points to the importance of dry, east winds (Agee & Flewelling, 1983) and specific surface airflow systems with offshore components that lead to fire weather (Heilman et al., 1994; Schroeder et al., 1962).

Potential Effects of 21st Century Climate Change on Fire Regimes

Even moderate (e.g., $\pm 0.3^{\circ}\text{C}$ decadal mean) fluctuations in PNW climate over the 20th century have influenced wildfire activity based on our analysis. Similar trends have been reported for other regions of the western U.S. (Westerling & Swetnam, 2003). Forest fire activity has been sensitive to climate variability, even in the face of altered dynamics due to fire suppression as in the case of our analysis using historic data. It is likely, therefore, that fire activity will increase over the next century in response to future temperature increases, at the same or greater magnitude as experienced during past climate variability. Forest fire activity in the western U.S., and in the northern Rocky Mountain region especially, may already have increased since the mid 1980s in response to warming and earlier spring snowmelt (Westerling, Hidalgo, Cayan, & Swetnam, 2006).

While some General Circulation Model (GCM) produced climate scenarios (e.g., Hadley Centre) predict increases in precipitation for the PNW over the next century, this is unlikely to offset wildfire risks because net summer soil moisture is also predicted to decrease (Keeton et al., in press). This is apparent in analyses integrating potential changes in temperature and precipitation. One such exercise used the MAPSS-Century vegetation change model (Neilson & Drapek, 1998), which includes a fire component. This model has been run using the HADCM2 and CGCM1 climate scenarios at monthly time steps from 1895 to 2100. Under both scenarios, the biomass consumed by fire in the PNW increases markedly by the middle of the 21st century (Bachelet, Neilson, Lenihan, & Drapek, 2001).

Ultimately, whether or not fire hazards increase will depend on changes in the intensity and duration of extreme fire weather; McKenzie et al. (2004) suggest that these will increase across the western U.S. in response to climate change. If the extreme drought conditions become more prevalent we can expect a greater frequency of large, high-intensity forest fires based on our results and previous research (Brown et al., 2004; Westerling & Swetnam, 2003). Increases in forest fire activity may be particularly pronounced in drought-prone portions of the interior PNW. In interior forests, increased

fuel production outside of summer, caused, for instance, by the increased spring and fall precipitation predicted by some climate models, could exacerbate wildfire risks associated with summer drought.

There is a rapidly emerging consensus that western wildlife risks, in both wildland areas and along the suburban–wildland interface, will increase substantially over the next century. Our predictions based on climate projections (Mote et al., 2003) are supported by studies that have simulated wildfire activity under altered climatic conditions (Brown et al., 2004; McKenzie et al., 2004; Price & Rind, 1994). Predictions of increased fire frequency, intensity, and extent under a doubled CO₂ climate are consistent across a range of regions, including the northern Rocky Mountains (Gardner, Hargrove, Turner, & Romme, 1996; Romme & Turner, 1991), temperate and boreal forests in Canada (Flannigan & Van Wagner, 1991), and northern California (Torn & Fried, 1992). Altered fire regimes are likely to cause related changes in both forest structure and species distribution patterns (Fosberg et al., 1992; McKenzie, Peterson, & Alvarado, 1996).

It is important to note that there are uncertainties in predictions of increased fire frequency and intensity with warmer, drier conditions (Agee, 1993). These include the difficulty of predicting potential changes in other important factors that influence fire activity, such as wind direction, synoptic-scale sequences of weather, and lightning activity (Agee, 1993; Agee & Flewelling, 1983). Uncertainties regarding lightning, however, may be overshadowed by the fact that the vast majority of fires in the Northwest are human caused; we found that over 80% of forest fires were human caused during the 20th century. The dynamics of natural ignition sources are thus unlikely to limit fire activity. If human ignition sources remain dominant and fire susceptibilities increase, fire activity will change regardless of lightning activity. It is uncertain whether societal changes, such as cultural, educational, or technological changes, might reduce or elevate future risks related to human caused ignitions.

Potentially elevated fire risks associated with climate change are compounded by the increased fire hazards created by 20th century fire exclusion in lower elevation forests of the interior Northwest (see Chapter 5 by Menning in this volume for further information on the history and consequences of fire suppression). In forests that once supported low to moderate severity fire regimes, fire suppression, logging, and grazing have increased susceptibilities to fire, insects, and pathogens by increasing stand densities and associated drought stress and by decreasing landscape heterogeneity (Hessburg, Mitchell, & Filip, 1994; Lehmkuhl, Hessburg, Everett, Huff, & Ottmar, 1994; Swetnam, Wickman, Paul, & Baisan, 1995). Consequently,

climate changes leading to increased frequency and intensity of summer drought could exacerbate the already elevated susceptibility of some interior Northwest forests to disturbances, particularly in dry forest types at lower elevations.

Climate change has additional implications for positive feedbacks between fire and other disturbances, such as synergistic interactions between fire and insect outbreaks. For example, outbreaks of bark beetles and other cambium-feeding insects are sometimes triggered by fires that weaken or kill trees (Agee, 1993; Hessburg et al., 1994). Similarly, increased frequency and intensity of insect attacks increase dead and dying fuel loads and associated fire hazards. Increased drought frequency would perturb physiological mechanisms involved in these feedback relationships, such as production of defensive compounds by trees. Multiple climate change-related stresses have the potential to create feedback loops that reinforce trajectories of change in disturbance regimes and related alterations of ecosystem structure and function.

Implications for the Urban–Wildland Interface

Predicted climate change impacts on forest fire hazards in the western U.S. raise a red flag for those engaged in fire management planning within the urban–wildland interface. Suburban or “exurban” (Theobald, 2005) expansion into partially or fully forested areas in fire-prone regions brings people and property into direct conflict with systems where fire is both natural and frequent (see Chapter 4 by Paterson in this volume for more information on fire-safe planning in interface communities). These threats have become far more pronounced in recent decades (Cova et al., 2004; GAO, 1999). For example, from 1980 to 2000 the area of suburban and exurban development increased by 133, 143, 117, and 124% for Idaho, Montana, Oregon, and Washington, respectively (Theobald, 2003a). In drier, fire-prone areas, suburban developments and scattered dwellings already face elevated fire risks due to the effects of past fire suppression. Increased vulnerability to forest fires over coming decades caused by climate change may worsen the current fire management problem. It is highly likely that this situation will become increasingly prevalent over the next century (Brown et al., 2004; Mote et al., 2003), necessitating further adaptation of both suburban development planning and allocation of fire management and restoration resources.

Forest managers are struggling to find politically acceptable, ecological sound, and financially expedient solutions to the current fire hazards, for

instance through prioritized fuels treatment and fire restoration projects. The goal typically is to restore forest stand and landscape-level characteristics associated with historic fire regimes. When these projects involve thinning forest stands or prescribed burning, support from local residents can vary widely (Brown et al., 2004; Pyne, Andrew, & Laven, 1996; Sagoff, 2004). For instance, the potential for prescribed fires to generate large volumes of smoke and associated reductions in air quality sometimes leads to public opposition (see Chapter 5 by Menning in this volume for further description of the social and regulatory constraints to prescribed burning). Fire restoration planning and implementation are highly contentious on federal lands, especially when they involve high levels of timber harvest and forest management in recreational or roadless areas. Forest managers must weigh sometimes competing management objectives: areas prioritized for fuels treatment do not always coincide with degree of threat to communities. For example, the fall 2003 fires that burned more than 774,000 acres in southern California (Fig. 7) were located primarily on non-federal lands (68%) and in non-forest vegetation (78%) according to an analysis conducted by The Wilderness Society (2003). Yet proposed federal fire

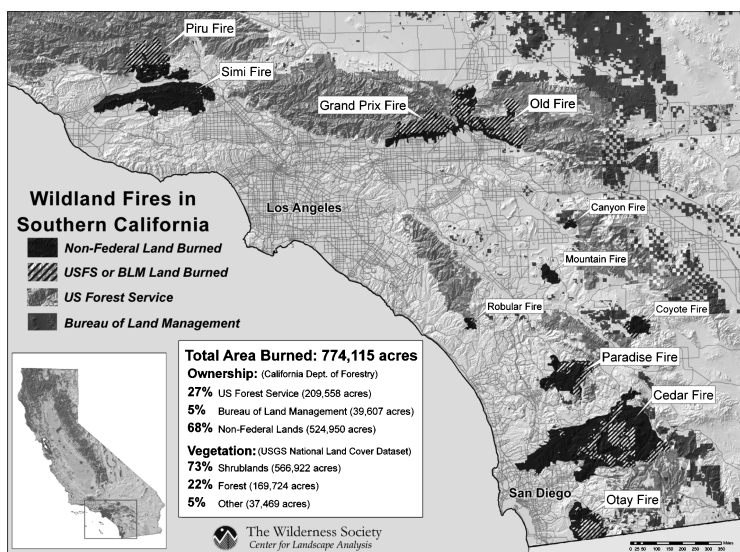


Fig. 7. Extent of the 2003 Wildfires in Southern California. Figure is reprinted with permission from The Wilderness Society (2003).

restoration projects target primarily federal lands, representing only 32%, and the more remote, forested periphery, of the burn area (The Wilderness Society, 2003). If fire-related threats to human safety and property increase as a result of climate change, fire restoration planning would need to be modified accordingly. For instance, with limited resources available, the location of projects may need to be prioritized based on: (a) degree of threat to human safety and property (e.g., proximity to semi-developed areas); and (b) landscape-level fire containment strategies, such as creation of fuel breaks and defensive zones.

Fire suppression in the U.S. currently costs over \$1.6 billion in annual expenditures (Whitlock, 2004), and may be as high as \$2 billion on average (Brown et al., 2004). Because hotter temperatures in the future will expand the duration of fire seasons (McKenzie et al., 2004), cautionary “no burn” periods, vegetation clearing around buildings, and maintenance of reserve fire fighting capacity will need to be extended as well. Between 1980 and 2000, 90% of fire suppression costs were accounted for by the most expensive 20% of fires exceeding 40 hectares (Brown et al., 2004). Consider that our results, as well as previous research (McKenzie et al., 2004), suggest that extreme fire weather is the dominant driver of fire vulnerability at regional scales. McKenzie et al. (2004) suggest that extreme fire weather will become more prevalent with climatic changes. This is likely to increase the intensity, or energy release component (ERC, a measure integrating precipitation and relative humidity and representing weather conditions as per the National Fire Danger Rating System), of future fires (Brown et al., 2004). The time period of ERC values above 60 (a threshold correlated with large fires) is predicted to increase by two weeks by 2070 across the western U.S., with the exception of Colorado, Montana, and Wyoming, where little or no change of this type is predicted (Brown et al., 2004). Hotter fires in the future will thus increase suppression and containment costs, disproportionately. Within the urban–wildland or exurban–wildland interface these costs may be even higher, relative to undeveloped areas, because of the need to protect human safety and property (Keeley, 2002).

The number of fires and area burned increases as population density increases (Keeley, Fotheringham, & Morais, 1999); we have found that the vast majority of fires during the 20th century in the PNW were human caused rather than ignited by lightning (Keeton et al., *In press*). Thus, if climate variability enhances conditions for fire spread, for instance through staggered periods of increased fuel production followed by drought (Brown et al., 2004; Keeton et al., *In press*; McKenzie et al., 2004; Swetnam & Betancourt, 1990), it is likely that the frequency and extent of uncontrolled,

human-caused fires will also increase in areas with human habitation and activity. For example, the increased occurrence of chaparral fires in southern California appears to be directly related to expanding human populations – and thus human-caused ignitions – rather than fire suppression or fuel loads (Keeley, 2002). While the dry, autumn Santa Anna or “foehn” winds are the primary cause of large, spreading shrubland fires in southern California (Keeley et al., 1999), susceptibility to wind-driven fires is predicted to increase with climate change in this region if drought conditions worsen (Torn & Fried, 1992). Thus, shrubland fire activity in southern California will reflect interactions between climate change effects and human-caused ignitions. Shorter fire-return intervals in savanna, shrubland, and chaparral systems enhance opportunities for the spread of weedy or annual species, including exotic species. This creates positive feedbacks with fire that contribute to the displacement of native species (various citations in McKenzie et al., 2004).

Adaptation of fire management and planning will be essential to address fire hazards in the urban–wildland interface as the climate changes. Development patterns in much of the U.S. have shifted fundamentally from what, in the past, might have been a clearly discernable expansion along an “urban–wildland interface” to what has become a mixed-use pattern of dispersed dwellings within a matrix of undeveloped vegetation (Brown et al., 2005; Theobald, 2005). This greatly complicates both fire restoration and fire fighting. In some cases public opposition (Sagoff, 2004) and fragmented ownership patterns make strategic planning of fuel breaks and buffer zones more challenging. During large fire outbreaks, fire fighting resources and public safety personnel must be reallocated to protect human life and property.

A comprehensive strategy to address this situation will require many elements (see, for example, Cohen & Saveland, 1997; Keeley, 2002; Lavin, 1997; Summerfelt, 2002); we will make three key recommendations. First, managers should consider focusing efforts on the strategic placement of fire restoration (Hardy & Arno, 1996; Raymond & Peterson, 2005) and fuel profile modification projects (i.e. “fuel breaks”) within and around the urban–wildland interface, to the extent this can be delineated given exurban development (see Chapter 3 by Stephens & Collins in this volume for a further description of urban–wildland interface zone fire containment strategies). The potentially negative effects of these projects, such as habitat fragmentation and vectoring of non-native species (Merriam, Keeley, & Beyers, 2006), also need to be considered carefully in determining where and if fuel breaks are desirable. Fire restoration should emphasize the

urban–wildland interface rather than previously unmanaged backcountry because: (a) this approach may be more cost effective (Canton-Thompson et al., 2006); and (b) higher-elevation forest types and unroaded backcountry are the least likely to have been affected by fire suppression (Bessie & Johnson, 1995; Quigley et al., 1996). Areas should be prioritized for treatment based on landscape-scale planning in this manner. Prioritization should consider possible fire behavior and spread, for instance using fire behavior models where applicable and feasible given input data requirements (see, for example, Scott & Burgan, 2005). It should also include pre-fire planning of fire-fighting strategies and tactics, personnel deployments, and evacuation routes (Rhode, 2004). Increased allocation of resources to this type of activity may prove more cost-effective and ecologically sound in terms of: (a) limiting the spread of fires; (b) protecting human property; and (c) restoring natural fire dynamics and ecosystem characteristics associated with historic disturbance regimes. Central to these efforts will be long-term planning in anticipation of future forest ecosystem change. This would require a significant change in forest management to consider climate information, which, based on our surveys of forest managers (Keeton et al., *In press*; Mote et al., 2003), is rarely considered.

Our second recommendation is to make better use of climate forecasts. Our current ability is limited to short-term predictions of fire weather. But improved understanding of PDO and ENSO will improve fire management planning with forecasted fire conditions anticipated over months to perhaps one or more years in advance (McKenzie et al., 2004). As relationships between variables used in current fire hazard prediction, such as the National Fire Danger Rating System, and modes of climatic variability are better documented, it will be possible to connect, in an increasingly sophisticated manner, fire prediction with climatic forecasts.

Finally, it is essential that communities carefully consider the consequences of continued residential expansion into fire-prone forested and shrubland areas. Apart from ecological concerns associated with sprawl and habitat fragmentation, risks from fire to property can represent an enormous financial burden on local communities, both in terms of insurance, pre-fire damage prevention, fire fighting, and repair following fires (Summerfelt, 2002; Truesdale (1995)). Climate change presents additional risks in terms of the potential for increased costs and threats in the urban–wildland interface. Growth management planning and open space conservation offer tools to minimize these impacts (Theobald, 2003b). These may prove especially fruitful if climate change affects forest fire regimes as we have predicted.

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