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Fire Severity in Conifer Forests of the Sierra Nevada, California

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Abstract

Natural disturbances are an important source of environmental heterogeneity that have been linked to species diversity in ecosystems. However, spatial and temporal patterns of disturbances are often evaluated separately. Consequently, rates and scales of existing disturbance processes and their effects on biodiversity are often uncertain. We have studied both spatial and temporal patterns of contemporary fires in the Sierra Nevada Mountains, California, USA. Patterns of fire severity were analyzed for conifer forests in the three largest fires since 1999. These fires account for most cumulative area that has burned in recent years. They burned relatively remote areas where there was little timber management. To better characterize high-severity fire, we analyzed its effect on the survival of pines. We evaluated temporal patterns of fire since 1950 in the larger landscapes in which the three fires occurred. Finally, we evaluated the utility of a metric for the effects of fire suppression. Known as Condition Class it is now being used throughout the United States to predict where fire will be uncharacteristically severe. Contrary to the assumptions of fire management, we found that high-severity fire was uncommon. Moreover, pines were remarkably tolerant of it. The wildfires helped to restore landscape structure and heterogeneity, as well as producing fire effects associated with natural diversity. However, even with large recent fires, rates of burning are relatively low due to modern fire management. Condition Class was not able to predict patterns of high-severity fire. Our findings underscore the need to conduct more comprehensive assessments of existing disturbance regimes and to determine whether natural disturbances are occurring at rates and scales compatible with the maintenance of biodiversity.

Key words: Condition Class; ecological restoration; Jeffrey and ponderosa pine; fire rotation interval; fire severity; fire spread; mixed conifer forests; spatial heterogeneity.

Introduction

The diversity of species in ecosystems is linked to natural disturbances and the environmental heterogeneity they create (Connell 1978; Huston 1979). However, managing the rates and scales of disturbance processes to allow for natural levels of environmental heterogeneity has its inherent risks and difficulties. This is particularly true for large disturbances that have profound influences on ecosystem structure, function, and composition (Turner and Dale 1998). Thus, although natural disturbances are vital to ecosystem integrity, maintaining their full range of variability is often at odds with management (Holling and Meffe 1996). How can disturbance-mediated environmental heterogeneity be most effectively maintained or restored where it has been suppressed over large areas? How can we recognize the levels and types of disturbance and heterogeneity that are appropriate for maintaining biodiversity? Here we explore these questions by focusing on the management of fire. Enormous resources are expended

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worldwide in efforts to manage this important disturbance or restore its effects.

To date there has been little direct assessment of how fire-mediated spatial heterogeneity might be restored or managed for in many fire-prone systems, such as the conifer forests of western North America (Rocca 2004). In many of these areas management policy is focused on the use of mechanical treatments to modify forest structure as a means of counteracting the effects of fire suppression. These efforts are controversial and are often not based on a sound understanding of the ecological role of fire as a disturbance process and the methods needed to restore its effects (Johnson 2003; DellaSala and others 2004). Perhaps nowhere in western North America has the appropriateness of structure-versus process-based forest management approaches been more controversial than in the conifer forests of the Sierra Nevada Mountains of California, USA (Stephenson 1999; Miller and Urban 2000).

Since the 1850s, grazing and fire suppression have reduced fire frequencies in the forests of the Sierra Nevada (Stephenson 1999; Miller and Urban 2000). The prevailing management view is that, because of fire exclusion, forest fires in the Sierra, which once varied considerably in severity, are now almost exclusively large, high-severity, stand-replacing events (Skinner and Chang 1996). As a consequence, an extensive program for the management of national forest lands was initiated in 2004. Its goal is to modify the structure of 283,000 ha of vegetation per decade, mainly in the dominant mixed conifer forests (USDA 2004). However, the actual severity of contemporary fire on these lands has yet to be analyzed to determine how well the prevailing view of dramatically increased fire severity and decreased heterogeneity is supported by empirical evidence.

Under the provisions of the National Forest Management Act of 1976, the national forests in the Sierra Nevada and throughout the United States are directed to ''provide for diversity of plant and animal communities.'' Natural variation and the maintenance of biodiversity in ecosystems can be assessed based on the concept of ecological integrity. ''Ecological integrity'' refers to ecosystem wholeness, including the occurrence of ecological processes such as natural disturbances at appropriate rates and scales to maintain natural levels of biodiversity (Karr 1991; Angermeier and Karr 1994). To determine the appropriateness of process-based versus structure-based management approaches for the maintenance biodiversity, we need to understand how ecological integrity is affected by contemporary fires. Thus, one of our primary objectives is to evaluate the rates and scales of contemporary fire as a disturbance process and assess their appropriateness in the context of ecological integrity.

To pursue this objective, we analyzed fire-severity data from the three largest fires that have occurred in the Sierra Nevada since 1999, accounting for most of the area burned over this time. These fires occurred in landscapes where timber harvest and silvicultural activities have been uncommon. After these burns, fire severity was classified by multi–US agency Burned Area Emergency Rehabilitation (BAER) teams. The BAER fire-severity data are derived from pre- and post-burn satellite and photo images and are used to map the effects of the fire on overstory vegetation canopy. We supplement these data with measures of ponderosa and Jeffrey pine mortality taken on the ground in areas of high-severity as defined by BAER. These pines have been harvested in many areas, and there is considerable interest in restoring their natural abundance (SNEP 1996). To gain further insight into the rates and scales of disturbance by fire under current management, we also evaluated temporal patterns of burning since 1950 in the broader landscapes in which the three fires occurred. Fire suppression has been mechanized in its current form since about 1950.

Another of our objectives was to evaluate the effectiveness of a national approach for the assessment of fire regimes and to discover how they have changed. The current basis for this approach, now used throughout the United States, is Fire Regime Condition Class (hereafter Condition Class), (Hann and Bunnell 2001); see also http://www.frcc.gov). It is an index that Estimates departure from reference conditions in vegetation, fuels, and disturbance regimes. In the national forests of the Sierra Nevada, Condition Class has been based on the number of fires estimated to have been exclude in the landscape due to fire suppression. Considerable research has revealed that historically Sierran forests were burned mostly by surface fire, but that this regime has decreased dramatically due to fire suppression (Caprio and Swetnam 1995; Skinner and Chang 1996). Condition Class predicts that these circumstances will lead to a dramatic increase in fire severity and place forest ecosystems at high risk losing key components due to fire (Hann and Strohm 2003).

A new approach to mapping departure from reference conditions, LANDFIRE, is currently under development (http: www.landfire.gov). In addition to Condition Class, it relies on the rapid assessment and mapping of wildland fuels to identify potential conditions that promote fire. The use of approaches that map departure from historic reference conditions in management is advancing rapidly. In the United States, 25 million ha have been identified for fuel treatments based on Condition Class (Brown and others 2004). Thus, it is especially timely now to evaluate the efficacy of approaches that map departure from historic reference conditions as a means of predicting fire severity.

Methods

Study Areas

The Sierra Nevada Mountains of California are a high-elevation (3000–4000 + m tall), 8-million-ha, north/south-trending mountain range (Figure 1, inset). They are forested primarily by conifer vegetation. We evaluated fire severity in the three largest burns in the Sierra since 1999—the McNally, Manter, and Storrie fires. Older fires lacked comparable fire-severity data in digital form. Smaller burns since 1999 in the main part of the Sierra occurred in areas that have been altered by past or recent timber harvesting and silvicultural activities. These effects were rare in the three burns we studied. The 2002 McNally and 2000 Manter fires occurred in close proximity in the southern Sierra (Figure 1), whereas the 2000 Storrie fire occurred in the northern Sierra near the southern Cascades (Figure 2). Together, these fires encompassed most of the area of Sierran conifer forest that has burned in the last 5 years, for a total of 49,917 ha. The McNally fire burned within the Sequoia National Forest from 22 July until 27 August 2002. The Manter and Storrie fires burned in 2000, the former from 7 July until 10 August and the latter from 17 August until 17 September. Weather initially conducive to fire spread, combined with rugged topography, enabled these fires to escape control and subsequently burn for 4–5 weeks under variable weather conditions. All three of the burns occurred in landscapes where most forests were not located within known, historic fire perimeters. In the McNally fire area, shrub ages indicate that fires had occurred there 125–150 years earlier in locations where there was no mapped record of fire (Keeley and others 2005).

Conifer forests typical of midelevations of the western Sierra (for a more detailed description of Sierran forests, see Rundel and others 1977) were abundant in the landscape that burned in the fires, particularly mixed or individually dominated forests of red and white fir (*Abies magnifica, A. concolor*); Jeffrey, ponderosa, and sugar pine (*Pinus jeffreyi, P. ponderosa, P. lambertiana*); and incense cedar (*Calocedrus decurrens*). These species are often mixed with a deciduous and an evergreen oak (*Quercus kellogii, Q. chrysolepis*). Trees in these forests are generally tall, with many overstory trees exceeding 40–50 m. Canopies are usually closed but can be open as a result of rocky substrata and other edaphic factors, particularly on granitic ridges. Open forests are mostly dominated by Jef-

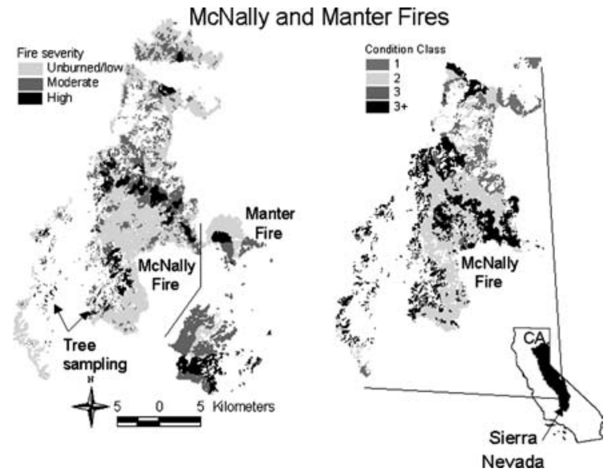

Figure 1. Patterns of burn severity in conifer-forested portions of the 2002 McNally and 2000 Manter fires in the southern Sierra Nevada, California. Preburn Condition Class is shown for the McNally fire area, not including the northernmost portion of the burn in the Inyo National Forest.

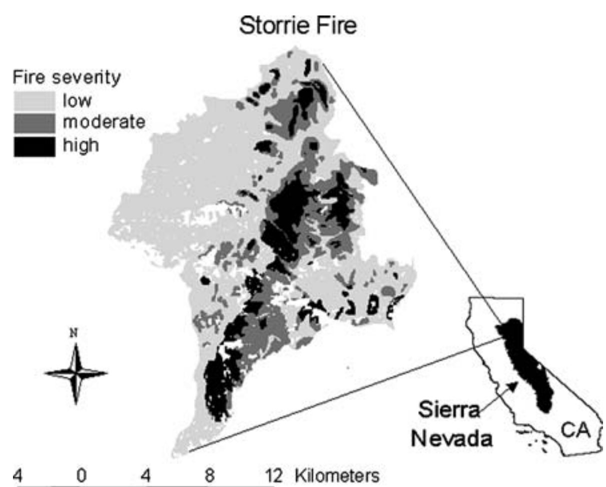

Figure 2. Patterns of burn severity in conifer-forested portions of the 2000 Storrie fire in the northern Sierra Nevada, California.

frey pine, often with shrubs in the understory. These forests are common in the Manter fire area and a portion of the McNally fire area. Closed mixed conifer forests predominated in the Storrie and McNally burn areas. One conifer, Douglas-fir (*Pseudotsuga menzesii*), is common in the Storrie fire area but absent from the southern Sierra.

Spatial Patterns of Fire Severity

BAER severity Mapping is designed to identify areas with high potential for soil erosion, which is generally based on the extent to which the fire affects the vegetation overstory canopy. The ability of remotely sensed data to identify patterns of fire severity based on the spectral response of tree canopies has been demonstrated in the Sierra (van Wagtendonk and others 2004). BAER severity in the McNally fire was mapped with Landsat 7 and SPOT multispectral satellite imagery (30-m pixel resolution) obtained immediately before and after the fire (Parsons 2002). A band ratio of mid-infrared and near-infrared reflectance was calculated from pre- and postburn image data. The band ratio data were classified and interpreted by staff at the USDA Forest Service Remote Sensing Applications Center in Salt Lake City Utah. BAER severity for the Manter and Storrie fires was mapped using aerial reconnaissance, infrared aerial photographs, and ground surveys (USDA 2000, 2002). General guidelines for severity classes are from the Forest Service Handbook (USDA 1995).

The BAER mapping identified three to four classes of fire severity based on the level of canopy effects detected. *Unburned* included areas where 0–10% canopy change was detected; this classification was distinguished only in the McNally fire. *Low severity* included areas where fire-caused crown scorch (heat-induced mortality of canopy foliage) affected less than 40% of overstory canopy foliage. The unburned and low-severity classes killed primarily conifer seedlings and saplings. *Moderate severity* included areas where fire scorched 40–89% of the forest canopy in the McNally fire and 40–80% in the other two fires. This level of severity was lethal to most conifer seedlings, saplings, and many small trees, but most overstory trees survived. *High severity* included areas where 90% or more of the canopy was scorched or affected by varying levels of incineration (direct consumption of crown foliage) in the McNally fire, whereas an excess of 80% of canopy showing these effects was considered high-severity in the Manter and Storrie fires. High-severity fire generally resulted in complete understory mortality. Overstory mortality ranged from complete to mixed depending on degree of canopy scorch and consumption (incineration), forest composition, and whether the threshold was 80% or 90% canopy mortality. Depending on imagery and other factors, different thresholds may be used for these severity levels in BAER mapping.

To characterize the spatial scales of the effects of high-severity fire in conifer forests, we describe the size of high-severity patches in each fire. To better characterize the effects, we evaluated the mortality of ponderosa and Jeffrey pine in areas of high-severity burn. Mortality assessments were restricted to a section of roadway in the McNally fire along which initial crown scorch had been assessed before there was any flushing of foliage. We identified five patches along this roadway that were dominated by trees that had no green foliage after the fire. These patches had fire effects ranging from 100% crown scorch (needles killed but not consumed) to needles consumed by crown fire. Within the patches, we chose to monitor all pines showing this range of high-severity effects that had a diameter at breast height (dbh) of more than 25 cm. These trees were generally within 50 m of the road. Our survival data are from 2 years postfire, following Stephens and Finney (2002). We did not observe any further indirect mortality caused by bark beetles over this period. Some trees were considered dead and were harvested over the course of the monitoring. We classified them as having been fire-killed, thus providing a maximum estimate of direct fire-induced mortality in the five sites.

Spatial and Temporal Patterns of Fire

To help assess the landscape-level influence of fire over time under modern fire suppression management, we calculated fire rotation intervals (amount of time needed for an area the size of the area of interest to burn one time) using fire perimeter data obtained from the US Forest Service and the California Department of Forest and Fire Protection. We used the total area of fire that has occurred from 1950 to 2005. Fire perimeters are complete and accurate over this period, and modern fire suppression was a consistent factor. Only conifer-forested areas were analyzed. The landscape we used to calculate fire rotation intervals in the McNally and Manter fire region was the southern portion of the Sequoia National Forest (210,932 ha of conifer forest), along with a smaller amount of the adjacent Inyo National Forest (10,000 ha of conifer forest), including and just beyond the northern boundary of the McNally fire (Figure 1). The landscape used to calculate fire rotation intervals in the Storrie fire region was the largest area within the Lassen and Plumas National Forests; that had the same forest vegetation types found within the Storrie fire region, which was in the center of this landscape. This landscape was more strongly dominated by conifer vegetation, which totaled 488,337 ha, than the landscape where the other two burns had occured. An estimate of rotation intervals for different severity classes in the two landscapes was calculated by assuming that all the conifer forest landscape that burned from 1950 to 2005 had the same severity proportions for the respective landscapes as either the McNally and Manter fires combined or the Storrie fire. This estimate integrates frequency and severity to help illustrate the influence of fire in the two landscapes under current management.

Fire Patterns and Condition Class

We evaluated fire patterns as a function of Condition Class in detail for the McNally fire, where preburn Condition Class data were available. These Condition Class data were mapped to the same vegetation units used in the Cal-Veg map (see Data Analysis). The Condition Class data were based on preburn Fire Return Interval Departure (FRID) and have been applied in planning efforts across the Sierra (USDA 2003, 2004). In other regions of the United States, the Condition Class approach is not necessarily based only on the estimation of FRID (http://www.frcc.gov). We obtained FRID data from the Southern Sierra Geographic Information System Cooperative, which helped to prepare them and still had a version that had not been updated after the McNally fire.

The Fire Return Interval Departure is the number of fires that, on average, may have been excluded. It is based on the time when fire last occurred in an area and the estimated historical fire frequency for the type of vegetation in that area. FRID was thus calculated as:

$$\text{FRID} = (Tsf - Fri)/Fri \qquad (1)$$

where Tsf equals time since the last fire in the landscape and Fri is the estimated fire interval for a vegetation type in the landscape. Estimated historical fire intervals for forests were developed from fire scar studies undertaken in the Sierra Nevada, southern Cascades, and the mountains of northwest and southern California, as reported by Skinner and Chang (1996). Table 1 shows estimated historic fire intervals for each forest type that burned in the McNally fire.

The FRID data we obtained identify the following categories of the number of fires that, on average, may have been excluded: *Extreme* denotes more than five (Condition Class 3 in the national three-level system), *High* is between two and five (Condition Class 3), *Moderate* is between one and two (Condition Class 2), and *Low* is less than one, or not outside the estimated historic fire return interval for a forest type (Condition Class 1) (USDA 2003). We kept the high and extreme FRID categories separate in our calculations and refer to extreme FRID as ''Condition Class 3+''.

Although preburn Condition Class data used in forest planning were not available for the same assessment in the Manter and Storrie fires, we make some inferences based on previous fire history, the Cal-Veg vegetation type within the burn perimeters, and the Condition Classes that would have been assigned based on the Condition Class criteria used in the Sequoia National Forest.

To determine how Condition Class might relate to fire spread rate—a likely predictor of fire severity that integrates weather, fuel, and topographic influences—we chose to assess BAER fire severity in relation to Condition Class in the McNally fire on days when the spread rate of fire was relatively rapid versus slow. To accomplish this, we plotted the ranked daily extent of total fire progression using data obtained from the Sequoia National Forest. This plot (Figure 3) shows that fire spread was particularly high on 2 days. Rather than analyze severity on just these 2 days, we selected additional days in which at least 2000 ha burned. On all the remaining days, an area equal to 1500 ha or less burned (Figure 3). The total areas on days where at least 2000 ha or 1500 ha or less burned were similar and constituted our relatively rapid- and slow-spread landscapes, respectively.

Data Analysis

We calculated fire-severity proportions in conifer forest vegetation types based on the primary vegetation type indicated in the vegetation map, Cal-Veg, that was used to develop Condition Class. It is a standard planning map used on national forest lands in California. Cal-Veg is a map representing current vegetation that is derived from satellite data. The map version used for the two fires in Sequoia had been updated just prior to the Manter fire, and the one for the Storrie fire had been updated the year before the fire. Updates were based

Table 1. Area of Different Conifer Forest Types Burned in the McNally Fire, Estimated Fire Interval used to Calculate Condition Class, and Percent BAER Severity for each Type

			Percent Fire Severity			
Type of Forest	Area (ha)	Fire Interval for Condition Class (y)	Unburned	Low	Moderate	High
Mixed conifer/fir	10,378	16	20.7	36.9	30.5	11.9
Red fir	10,323	50	38.6	35.1	16.3	10.0
Mixed conifer/pine	4154	16	5.5	33.5	52.1	9.0
Jeffrey pine	39,341	50	5.9	23.5	49.0	21.6
Ponderosa pine	2455	6	9.8	38.6	44.0	7.6
Lodgepole pine	1559	163	49.5	39.7	10.7	0.0
Subalpine conifers	692	163	28.9	60.8	9.9	0.4
White fir	117	16	14.9	47.0	34.4	3.6
Foxtail pine	92	163	70.5	29.5	0.0	0.0
Totals	33,704		23.4	35.1	30.5	10.9

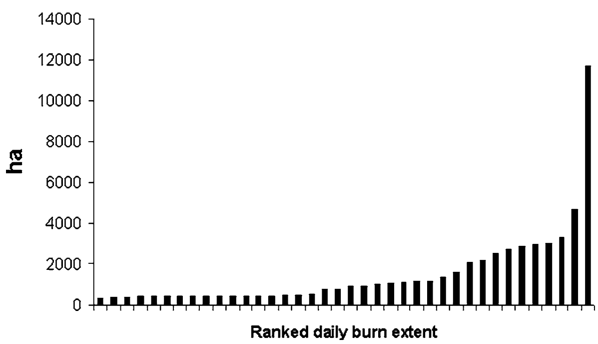

Figure 3. Ranked daily burn extent in the McNally fire as determined from the fire progression data of the Sequoia National Forest.

on accuracy assessments. A detailed description of the Cal-Veg map, and its development and accuracy for Forest Service lands, is at http://www.fs.fed.us/r5/rsl/projects/mapping. The minimum mapping unit is 1 ha. A description of the forest vegetation alliances mapped for the southern and northern Sierra and described in the results can be accessed at http://www.fs.fed.us/r5/rsl/projects/classification/zone-map.shtml.

We excluded pinyon/juniper woodlands and a small amount of open forest on the more arid east side of the Manter fire because it was not in national forest land and was subjected to different mapping protocols. Conversely, we included a small amount of area where the vegetation map indicated a hardwood conifer mix, but where the primary dominant was a conifer forest tree.

A formal statistical approach to testing for differences in severity proportions among Condition Classes by resampling independent, random point locations was not possible (for example, Odion and others 2004) because there was only enough area in some classes to locate a small number of independent points. Therefore, we present the proportions of fire severity by vegetation type and Condition Class and generally evaluate the weight of evidence provided by this information and other descriptors of the current fire regime in the context of the objectives described in the introduction.

Tree mortality was assessed for two diameter-size classes, 25–50 cm and larger than 50 cm. These two classes were compared for differences using a chi-square 2 × 2 independence test of the hypothesis that smaller trees would suffer greater mortality.

Results

Spatial Patterns of Fire Severity

Most of the conifer forests that burned in the McNally fire (Figure 1) showed characteristics of moderate- or lower-severity fire. High-severity fire accounted for 10.9% of all forest area (Table 1). The highest percentage of high-severity fire occurred in forests dominated by Jeffrey pine (22%), a species that is common on relatively dry and wind-exposed ridges. Most Jeffrey pine forest (83%) burned on the 3rd, 4th, and 5th most extreme-spread days of the McNally fire. Other forest types had much less high-severity fire—in particular, ponderosa pine, mixed conifer/pine, and the relatively small area of forest with long intervals of natural fire (mixed subalpine conifers, lodgepole pine, and foxtail pine). Although the McNally fire burned mostly fir and mixed conifer forests, most of the area that burned in the Manter fire was Jeffrey pine forest. The conifer forests in the Manter fire had more high-severity fire (29%) (Table 2). However, the Manter fire also had a lower

Table 2. Area of Different Conifer Forest Types Burned in the 2000 Manter and Storrie Fires, and the Percent BAER Severity for each Type

			Percent Fire Severity		
	Forest type	Area (ha)	Low	Moderate	High
Manter fire	Jeffrey pine	5,508	24.5	43.6	31.9
	Mixed conifer/fir	1,145	31.9	50.3	17.8
	Red fir	162	68.1	31.9	0.0
	Lodgepole pine	15	0.0	26.7	73.3
Totals		6,829	26.7	44.4	28.9
Storrie Fire	Mixed conifer/fir	7,583	85.8	10.0	4.2
	Mixed conifer/pine	6,577	45.6	26.3	28.1
	Douglas-fir/ponderosa pine	2,986	54.2	35.9	10.0
	White fir	1,511	72.6	5.9	21.6
	Red fir	591	95.8	2.4	1.8
	Jerey pine	128	41.7	52.8	5.6
	Lodgepole pine	7	100.0	0.0	0.0
	Ponderosa pine	2	100.0	0.0	0.0
Totals		19,384	66.3	19.2	14.5

threshold for high-severity fire than the McNally fire (80% versus 90% or more canopy foliage mortality).

For the Storrie fire, severity mapping also used the 80% threshold for high-severity fire. High-severity fire totaled 14.5% among all conifer forests, but the area incurred only about half as much moderate-severity fire as the area burned by the other two fires and consequently considerably more low-severity fire (Figure 2 and Table 2). Of the total area that did burn at high severity (2805 ha), most (1730 ha) of this fire occurred in mixed conifer/pine forests. However, forests dominated by ponderosa and Jeffrey pine had little high-severity fire. Conversely, white fir forests incurred much more high-severity fire than mixed conifer/fir, the most common forest type in the Storrie burn area. Thus, this fire had lower overall severity than the others, and even in different areas mapped with forest types that included many of the same species, the fire nonetheless burned with varying severity.

A few large high-severity patches accounted for much of the total area of high-severity fire in the conifer forests affected by the three burns (Figure 4A–C). However, all three fires produced mostly relatively small patches of high-severity fire. Patches totaling less than 5 ha accounted for 107 of the total of 157 high-severity patches in the McNally fire. They accounted for 28 of a total of 40 in the Manter fire, and 59 of 102 in the Storrie fire.

Many of the pines we monitored that incurred severe burn effects nonetheless produced new foliage from surviving terminal buds in the year after the fire. All surviving trees had either 100% crown scorch and no incineration of foliage or 100% scorch and incineration extending upward to at most 50% of total tree height. For Jeffrey pines incurring these fire effects, 22 of 44 trees survived and there was no difference between the 25–50 cm and greater than 50-cm diameter size classes in terms of the percentage of trees that survived. For the more abundant ponderosa pine, 42 of 83 and 57 of 83 trees in these two size classes survived, and diameter exerted a significant, positive effect (chi-square = 5.6, $P < 0.01$). None of the trees ($n = 90$) with higher levels of crown incineration, survived, indicating that there are significant differences between the effects of crown fire that incinerates foliage and the effects of severe surface fire, which primarily results in the death of foliage due to heat scorch.

Spatial and Temporal Patterns of Fire

For the larger landscape of the national forest in which the McNally and Manter fires occurred, the rotation interval from 1950 to the present for all fire was 185 years. The McNally and Manter fires were responsible for two-thirds of the area that was burned over this time. For both burns combined, the overall percentage proportions of high- and moderate-severity damage in conifer forests was 14% and 33%, respectively. Using these values, the rotation interval in conifer forests was estimated to be about 1330 years, for high-severity fire and about 565 years for moderate-severity fire. Fire has been less common in conifer forests of the Storrie

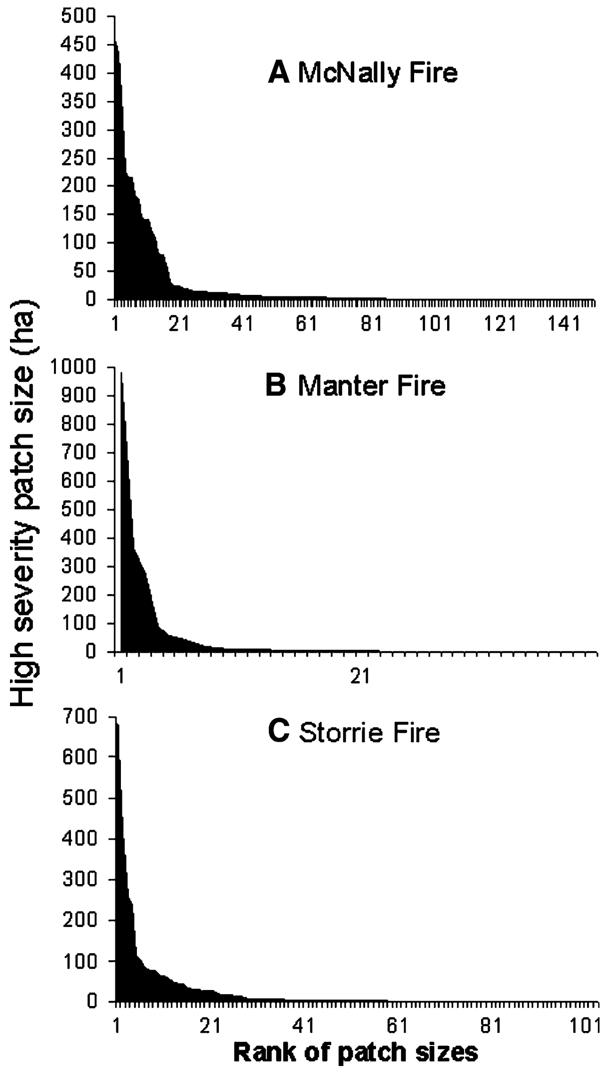

Figure 4. Ranked size of high-severity burn patches in conifer vegetation in the **A** McNally, **B** Manter, and **C** Storrie fires.

fire region. The rotation interval for all fire since 1950 was 507 years. The Storrie fire accounted for about half of all fire in conifer forests over this time period. The estimated rotation interval since 1950 was 3503 years, for high-severity fire and 2460 years for moderate-severity fire in the region in which the Storrie fire occurred.

Severity Patterns and Condition Class

Fire severity proportions by Condition Class under slow- and rapid-spread days in the Sequoia National Forest portion of the McNally fire are shown in Figure 5A–B. The 3939 ha comprising Condition Class 1 forests (2505 ha on slow-spread days plus 1424 ha on rapid-spread days) had almost no high-severity fire. These forests were predominantly comprised of subalpine and other high-elevation forests of red fir, lodgepole pine, and foxtail pine that grow on the relatively flat Kern Plateau.

For Condition Classes 2, 3, and 3+, there were distinctions in degree of severity between rapid- and slow-spread days. In particular, on rapid-spread days, moderate-severity fire was considerably more common, whereas low-severity was less common. The largest area of high-severity fire occurred on rapid-spread days in Condition Class 2 forests (Figure 5A). These forests were comprised mainly of red fir (62%) and Jeffrey pine (22%). Condition Class 3 forests consisted entirely of mixed conifer/fir or pine, whereas Condition Class 3+ forest were ponderosa pine. They had the same proportions of high-severity fire (13%) on rapid- and slow-spread days. This figure was very similar to that for conifer forests throughout the area covered by Condition Class data (Figure 1), which was 11.8%. Condition Class did not appear to have a strong effect in promoting rate of spread because a considerable area of Condition Class 3+ forest burned on slow-spread days (Figure 5).

Applying the McNally Condition Class criteria to the Manter burn area, we find that the 5400 ha of Jeffrey pine and 1145 ha of mixed conifer/fir forests that had no record of previous fire would be Condition Classes 2 and 3+, respectively. Jeffrey pine had 32% high-severity fire, and mixed conifer/fir forests had 17% high-severity. A small area of Jeffrey and lodgepole pine forest (94 ha) that would have been Condition Class 1 had 43% high-severity fire.

Applying the McNally Condition Class criteria to the Storrie fire area and presuming Douglas-fir/ponderosa pine to have an estimated past fire return interval of 16 years, like similar forests (Table 1), we find that there were 792 ha of Condition Class 2 mixed conifer forests. Most of this are burned previously in the 1970s and was primarily forested by Douglas-fir/ponderosa pine. In the Storrie fire, these forests burned with 20% high-severity and 53% moderate severity. Red fir and Jeffrey pine forests (719 ha) had no record of previous fire and would also have been Condition Class 2. They burned at much lower severity than most forests (Table 2). The rest of the forests affected by the Storrie fire had not burned for a long time and would have been condition Class 3+. Collectively they experienced the same severity proportions observed for the burn as a whole—lower than that seen in the Condition Class 2 mixed conifer forests.

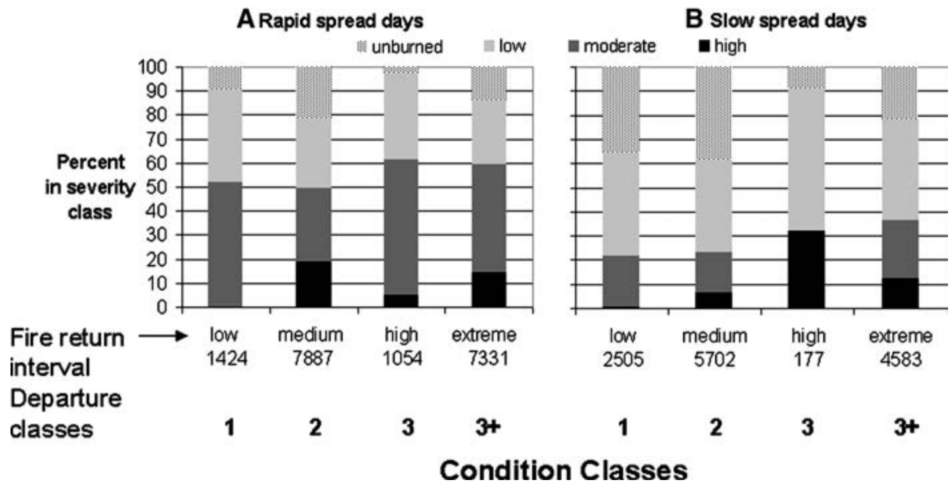

Figure 5. McNally fire severity proportions by Condition Class occurring during **A** days of relatively rapid fire spread (n = 10) and **B** days of relatively slow spread (n = 28). Numbers below columns are hectares burned.

Discussion

Contemporary fire is clearly not almost exclusively high-severity and stand-replacing in long-unburned areas of Sierran conifer forests. In the large area of burned forest that we evaluated, fire severity was highly variable and caused a relatively small amount of high-severity effects. Van Wagtendonk and others (2004) found similar levels of variation and severity proportions in another recent Sierra Nevada burn in the same forest types examined in our study. Our findings are also consistent with the result of recent modeling, which showed that long-unburned Sierran forests unaffected by silvicultural activities would not incur crown fire until temperature, relative humidity, and wind exceeded the 97.5th percentile of their summertime levels (Stephens and Moghaddas 2005).

The burn patterns we observed are also consistent with descriptions and evidence in Sierran forests not influenced by fire suppression and silviculture. There are a number of historical accounts of variability in fire ranging from light understory burning to patchy high-severity fire in Sierran mixed conifer forests, including one by the famed naturalist John Muir (reviewed by Stephenson and others 1991; Stephenson 1999), and another by a forest surveyor John Leiberg (1902). Recent studies using historic photos and field sampling have concluded that patches of high-severity fire have shaped mixed conifer forests in the Sierra Nevada and the adjacent southern Cascades (Russell and others 1998; Beaty and Taylor 2001; Taylor 2002). Show and Kotok (1924), Russell and others (1998), Beaty and Taylor (2001), and Taylor (2002) describe historic high-severity burn patches in the Sierra that are comparable in size to many of the larger patches produced by the three fires we studied. Smaller patches or gaps have also played an important role in determining forest and landscape structure and composition (Stephenson and others 1991; Keeley and Stephenson 2000) and were common in the three fires we studied. Leiberg (1902) and Beaty and Taylor (2001) have also describe the occurrence of large historic fires.

Because the fires we studied burned for 4–5 weeks, mainly in July and August, they were influenced by a range of weather conditions. This may help to explain why they were heterogeneous and qualitatively similar to descriptions of pre–suppression era fires. Most lightning ignitions occur in the Sierra during July and early August (Caprio and Swetnam 1995). Historic lightning ignitions that led to spreading fires would have been driven by the same seasonal patterns of warm, dry weather that typifies the Sierran summers. The large size of the fires we studied likely enhanced their variability by creating both fire-generated winds, which that can make combustion more active, and dense smoke, which can lower temperatures and mitigate fire behavior (Pyne 1984). Thus, it is important to stress that our results apply to fires in the Sierra that burn for long durations and spread over relatively large areas in mid- and late summer. These circumstances are representative of much of the areas burned by contemporary fire, and presumably fire in the past, given the effect of large fire on the cumulative amount of area burned. Much less heterogeneity may result from fires that burn for a shorter time and cover small areas. Our results also apply only to areas in the Sierra where timber harvesting and silvicultural activities have not been common. There are many areas of the Sierra that have been modified considerably by intensive silvicultural activities (SNEP 1996) and where severity is expected to be higher due to increases in available fuel and the loss of fire-resistant trees (Stephens and Moghaddas 2005).

After a long period of reduced fire influence, large, heterogeneous fires can hasten ecological restoration (Baker 1992; Miller and Urban 2000; Fulé and others 2004). They may affect biodiversity by thinning trees and decreasing competitive exclusion processes and by increasing structural and landscape diversity. Fire-created gaps provide opportunities for the natural regeneration of light-demanding conifers such as pines and giant Sequoia (*Sequoiadendron giganteum*) (Stephenson and others 1991; Keeley and Zedler 1998; Stephens and others 1999) whose natural abundance in the Sierra has been reduced (SNEP 1996). There are concerns about the lack of natural regeneration in these species due to the absence of fire severe enough to create openings, consume sufficient duff and litter to facilitate successful germination, and open cones in giant Sequoia (Stephenson and others 1991; Stephens and others 1999). Such fire effects may not only promote the natural reproduction of these conifers, but also favor the relative abundance of these species because they have a greater ability to survive. Large giant Sequoia may survive in areas of crown fire (Stephenson and others 1991), and we found that many medium and large ponderosa and Jeffrey pines can survive severe surface fire. There may be some additional mortality among these trees, but those that survive are likely to experience rapid growth and increased vigor, much like giant Sequoia after severe fire (Stephenson and others 1991). Mature white fir may also be more fire resistant in the Sierra than previously suspected, aided by their ability to produce epicormic branches (Hanson and North 2006). Surviving conifers may serve as sources of seed that help to ensure natural regeneration in high-severity burn patches.

Patches of habitat created by high-severity fire, with their rich array of snags, logs, and nonarborescent vegetation, are among the scarcest habitats in many forested landscapes (Lindenmayer and Franklin 2002). After 50–100 years this early successional habitat can succeed to forest (Russell and others 1998). Thus, based on estimates the area of high-severity fire predicted by our fire rotation analyses for the period since 1950 in the Sequoia and Storrie fire regions, about 4.2% and 1.5% of these landscapes, respectively, may have naturally developed early successional burned forest habitat under the current fire regimes. The maintenance of this habitat in the landscape by fire promotes biodiversity because it supports plant, insect, and wildlife assemblages not found in other Sierran habitats. In addition, there are numerous plant and animal species that have become rare due to their requirements for burned forest habitat. For example, there is some concern over the local extirpation of avian species with these habitat requirements (Kotliar and others 2002). Species such as the black-backed woodpecker (*Picoides arcticus*) may be indicators of whether sufficient, severely burned forest habitat is being maintained for biodiversity (Hutto 1995). These birds require young, severely burned patches of at least 12–25 ha (Saab and others 2002). The three fires we studied created 70 severe-burn patches larger than 12 ha where there had been none or very few due to the lack of fire.

Thus, the effects of the large fires we studied are consistent with the diversity goals of the National Forest Management Act. Elsewhere in the western United States, a number of large fires have also been found to perform the desired ecological functions of fire (for example, Turner and others 2003; Kotliar and others 2003; Fulé and others 2004; Odion and others 2004; Schoennagel and others 2004; Smucker and others 2005). These specific effects may ultimately be necessary for maintaining biodiversity that depends on fire. Prescribed burning can help, but it is limited in extent, severity, and heterogeneity (Baker 1994; Rocca 2004) and may not mimic natural fire (Moritz and Odion 2004). On National Forest Service lands, prescribed burning is often conducted outside the normal fire season, when flaming is subdued but wildlife such as herptofauna are highly vulnerable to smoldering combustion (Bury 2004). Neither these fires, nor the structural modification of forests through mechanical treatments, may provide fire-specific effects for species that require them (for example, flowering plants with fire-dependent seed germination that is sensitive to burn season, conifers with heat-opened cones, and cavity-nesting species that dependent on standing dead trees for nesting and foraging).

Fire Patterns and Condition Class

We found that the proxy for fire suppression effects, Condition Class, was not effective in identifying locations of high-severity fire. Condition

Classes 2, 3, and 3+ in the McNally fire all had similar fire severity proportions. When the same Condition Class criteria were applied to the other two fires, we found that fire severity generally decreased rather than increasing from Condition Class 2 to 3+. In short, Condition Class identified nearly all forests as being at high risk of burning with a dramatic increase in fire severity compared to past fires. Instead, we found that the forests under investigation were at low risk for burning at high-severity, especially when both spatial and temporal patterns of fire are considered.

The lack of an observed relationship between Condition Class and fire severity suggests that exogeneous forces such as weather, climate, topography, and neighboring vegetation (for example, pyrogenic shrubs) largely determine fire-severity patterns in forests. Because fire severity did not increase above Condition Class 2, the combustibility of Sierran forests may reach a maximum at the fire-free intervals indicated by this class (32–48 years for many forest types), (Table 1).

A number of interrelated factors may explain why these forests reach a maximum in combustibility. For example, the total leaf area of a forest reaches a maximum (Waring and Schlesinger 1985). Once forest overstories close in the Sierra, they may exclude pyrogenic shrubs with high light requirements (Show and Kotok 1924), greatly decreasing the potential intensity of understory combustion. The base height of the forest canopy sufficiently dense to propagate fire may also become relatively high in long-unburned forests (Stephens and Moghaddas 2005). In terms of surface fuel beds, those associated with Sierran conifers that increase in abundance with time since fire (for example, fir) are more dense than those found under pine and thus less combustible (van Wagtendonk and others 1998).

Conclusions

Our findings suggest that elevated risk of high-severity fire due to the effects of fire suppression is not the pervasive, predictable ecological problem that it has often been portrayed to be in the Sierran forests we studied. In addition, they provide evidence that fire alone can restore its past influence as a patchwise and stand-thinning disturbance agent as well as a facilitator of species diversity and fire-adapted conifers in these forests. Thus, it appears that management can shift toward process restoration by introducing more fire and increasing the use of wildland fire (Miller 2003). There may be no other effective strategy for restoring and maintaining ecological integrity and for fostering the natural diversity of species dependent on effects specific to fire. The structural modifications of forests cannot mimic the heterogeneous effects of fire. Instituting a policy that allows more fire to burn would require considerable planning and additional efforts to improve human safety, but such efforts are needed under any management scenario.

Both Condition Class and the new LANDFIRE approach are based on mapping any departure in fire regimes from reference conditions. Presuppression reference conditions for fire must be based on retrospective studies. These studies are too methodologically limited to provide a comprehensive description of the spatial extent and variation in the effects of past fires (reviewed by Veblen 2003). As a result, the importance of past surface fire may be overestimated and conversely, past heterogeneity in fire may be underestimated (for example, Minnich and others 2000). To add to the problem of uncertainty about past fire, there may be significant misconceptions about current fire severity that lead to further overestimation of the differences between past and present fire regimes.

By directly assessing existing fire regimes in the context of ecological integrity, we can avoid some of the problems that may arise when current methods for estimating departure in fire regimes are used. A general approach based on the assessment of existing rates and scales of processes in the context of ecological integrity has been recommended for the management of biodiversity as a means of overcoming problems in defining the ''natural'' range of variation in ecological systems (Parrish and others 2003). The direct assessment of fire regimes can be improved by applying more sophisticated mapping of fire severity and performing landscape analyses that provide a clearer link between pattern and process (Wagner and Fortin 2005). In the Sierra Nevada, it is important to distinguish high-severity surface fire from crown fire because the two types of behavior may have very different effects on tree mortality. There is also a need for analyses of fire behavior in areas affected by timber harvesting and silviculture. Finally, better integration of the spatial and temporal components of other forest disturbances in the Sierra Nevada in addition to fire, is needed to determine if their rates and scales are compatible with ecological integrity.

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