Quantifying the Consequences of Fire Suppression in Two California National Parks

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Excluding fire can have untold ecological effects. Decades of fire suppression in national parks and other protected areas have altered natural fire regimes, vegetation, and wildlife habitat (Chang 1996; Keane et al. 2002). Management actions to suppress lightning-ignited wildfires removes one of the most important natural processes from fire-dependent ecosystems, and yet resource specialists currently have no way of measuring or monitoring the effects of these actions.

Yosemite and Sequoia–Kings Canyon National Parks in California have been leaders in the restoration of fire as a natural process. By 1970, both parks had instituted a policy whereby lightning-caused fires could be allowed to burn in certain areas of the park, a strategy that is now known as "wildland fire use" (van Wagtendonk, in press; Kilgore 2007). Despite these efforts, the parks continue to struggle with restoring natural fire regimes, and the majority of lightning-caused ignitions are suppressed for a myriad of biophysical and social reasons. Concerns with allowing fires to burn include the risk of fire leaving jurisdictional boundaries, the potential that unnaturally high fuel accumulations and tree densities could cause unnatural and undesirable fire effects, and the impact of smoke emissions on surrounding communities. Management-ignited prescribed fire has been used both as a restoration tool and a means to mitigate the risk of severe fire (Keifer et al. 2000a).

To help prioritize prescribed fire and other restoration activities, Yosemite and Sequoia–Kings Canyon National Parks use the Fire Return Interval Departure (FRID) Index to quantify departure from the fire return interval that existed prior to Euroamerican settlement (Caprio et al. 2002; van Wagtendonk et al. 2002). FRID is computed as the amount of time since the last fire (time-since-last-fire) divided by the characteristic fire return interval for the vegetation type. The characteristic fire return interval can be determined from published literature (Fischer et al. 1996) and/or fire history chronologies reconstructed from the tree rings of fire-scarred trees (Caprio and Lineback 2002). Through the use of geographic information system (GIS) software, FRID has been spatially mapped, and areas with the highest values, or "ecological need," are typically prioritized for fuel management and restoration activities (Caprio et al. 2002). FRID is also useful as a coarse filter for measuring progress and setting maintenance priorities in ecological restoration, with a decrease in FRID values reflecting improved ecosystem condition (Caprio and Graber 2000). Median or mean fire return intervals are typically used to calculate FRID, although "average" maximum fire return intervals have been used to generate conservative estimates of FRID (Keifer et al. 2000b).

A plethora of computerized models and tools are available to support fire management planning (Stratton 2006). One of those is FARSITE, a model that uses spatial information on topography and fuels along with weather and wind data to simulate the spread and behavior of wildland fire (Finney 1998). Although FARSITE's predictions are most commonly used to support fire incident management, it can also be used to investigate where fires in the past *might* have spread had they not been suppressed. This retrospective application is particularly appealing because it avoids the uncertainty inherent in weather forecasts. When applied to past events, actual weather observations can be used as input to FARSITE.

We used retrospective fire behavior modeling and the FRID index to quantify the consequences of suppression. We conducted analyses for case study watersheds in Yosemite and Sequoia–Kings Canyon National Parks. To our knowledge, this is the first time case studies have been used to systematically evaluate the consequences of suppression decisions. We believe the approach could be adapted for application elsewhere. A forthcoming report describes methods in detail (Davis and Miller, in preparation).

Methods

Study areas Historically, at least 6,500 ha in Yosemite National Park (Yosemite National Park 2003) and 10,000 ha in Sequoia–Kings Canyon National Parks probably burned each year (Caprio and Graber 2000). Burning by Native Americans contributed to the historical fire frequency in both parks, but probably only in certain areas (Vale 1998). From 1930–2000, wildland and prescribed fires only burned an average of less than 1,250 ha per year in Yosemite (van Wagtendonk et al. 2002) and 855 ha per year in Sequoia–Kings Canyon (Caprio and Graber 2000). Changes resulting from the lack of fire are most pronounced in the lower elevations of both parks' frontcountry with oak woodlands, ponderosa pine, and mixed conifer forests. Twenty-five percent of the vegetation in Yosemite and 22% in Sequoia–Kings Canyon is considered to be in a state of high departure from natural conditions, as defined by high FRID values. Most of this area occurs in the lower-to-mid-elevation conifer forests.

We selected the 31,400-ha South Fork Merced (SFM) watershed in Yosemite and the 90,700-ha Kaweah watershed in Sequoia–Kings Canyon for our retrospective analyses (Figure 1). Most of the SFM watershed last burned prior to the 1930s. Areas of special concern include the townsite of Wawona and the Mariposa grove of giant sequoia trees (*Sequoiadendron giganteum*). Fires are typically suppressed in this area, which has led to unnaturally high fuel accumulations. The Kaweah watershed in Sequoia–Kings Canyon contains most of the park's infrastructure, most of its giant sequoia groves, and has a diversity of boundary interface issues. Due to its proximity to developed areas and topography that drains into the San Joaquin Valley, smoke and its impacts on air quality are a primary concern. About half of the lightning ignitions in the watershed are suppressed.

Data Retrospective fire behavior modeling requires high-quality ignition, weather, and fuels data. A combination of improved record-keeping on fires by the parks, the implementation of national fire planning and budgeting analyses, and the use of remote-sensing data have provided datasets of adequate quality starting in 1994. The study period for our retrospective analyses was 1994–2004.

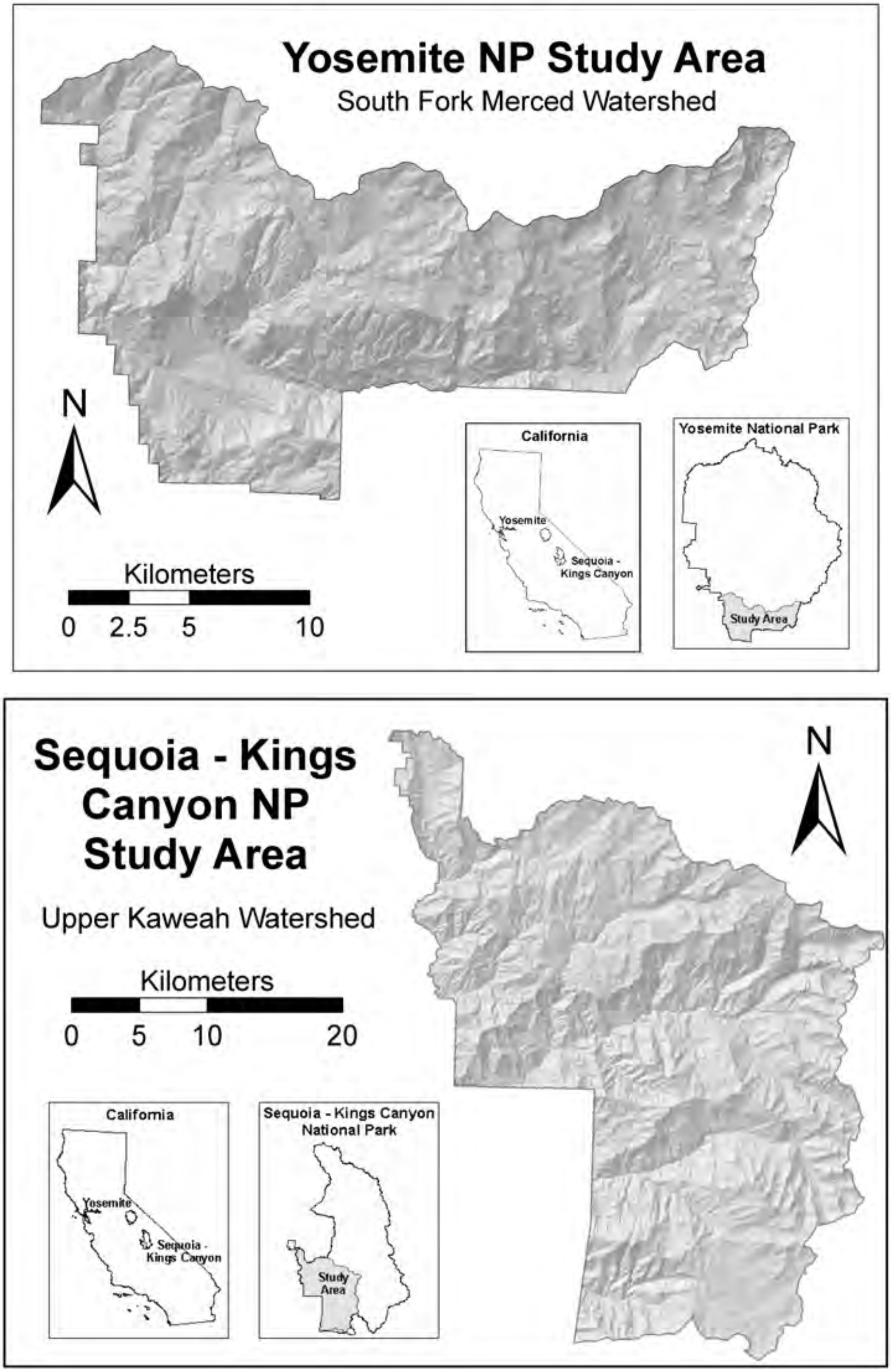

Figure 1 The 31,400-ha South Fork Merced (SFM) watershed in Yosemite National Park and the 90,700-ha Kaweah watershed in Sequoia–Kings Canyon National Parks.

We used historical wildland fire ignition data from the parks to identify lightning ignitions that were suppressed during the study period. Data attributes included location, start date, cause, management response, and final fire size.

Weather data recorded by representative meteorological stations in or near the study area were used to select ignitions with the potential for significant growth and to reconstruct the conditions under which we modeled the spread of the selected ignitions. Hourly data from Remote Automated Weather Stations (RAWS) spanning the 11-year study period were obtained from the Western Regional Climate Center (WRCC 2007) and used as input to the fire spread simulations. Daily RAWS data over a period of 17 years (1991–2006) for the SFM watershed and 31 years (1973–2004) for the Kaweah watershed were correlated with actual fire activity to identify conditions that support active fire spread. These correlations were used to develop rules for selecting which ignitions to model, and for identifying simulation days for the fire behavior simulations.

Vegetation data were previously derived by each of the parks from satellite and aerial imagery. To generate necessary spatial input data for fire spread modeling, the vegetation data were “crosswalked” to 22 distinct fuel types represented by surface-fire behavior fuel models (Scott and Burgan 2005). Fire behavior fuel models serve as composite descriptions of several fuelbed inputs needed for surface-fire behavior modeling by FARSITE (Stratton 2006).

Topographic data were required for the fire behavior simulation. Elevation, slope, and aspect data were obtained from the parks at 30-m spatial resolution.

Historical fire perimeters were available as digital fire atlases for the period 1930–2004 for the SFM watershed, and for 1921–2004 for the Kaweah watershed. These fire atlas data were used in retrospective modeling to update spatial fuel data between simulation years and to modify the fuels data during a fire simulation. These data were also used to map the time-since-last-fire for the computation of FRID. Burn severity data derived from satellite imagery were available for eight of the real fires in the SFM watershed and two in the Kaweah watershed during the period of the study, and were used to update the fuels during the course of the analysis (Thode 2005; Miller and Thode 2007).

Models We used a dynamic model of fuel succession to represent fuel accumulation and post-fire effects during the 11-year study period. This expert-opinion-based fuel succession model was developed in collaboration with scientists and managers from the parks and the US Geological Survey (USGS). It is a deterministic model that predicts how fuels—represented by one of 22 fire behavior fuel models—can be expected to change over time. Transitions from one fuel model to another and the rates of these transitions were based on expert knowledge of how quickly fuels accumulate in the associated vegetation types and how that vegetation would be expected to react to fires of low, moderate, or high burn severities. We defined burn severity according the degree of fuel consumption that would be seen from a remotely sensed (aerial) perspective. Twenty-two diagrams were created to describe fuel succession for each of the fuel models present in the parks (Figure 2).

We used the computer simulation model FARSITE (Finney 1998) to determine where fires might have spread had they not been suppressed. Spatial data input to FARSITE were topography (elevation, slope, aspect), fire behavior fuel models, canopy cover, canopy

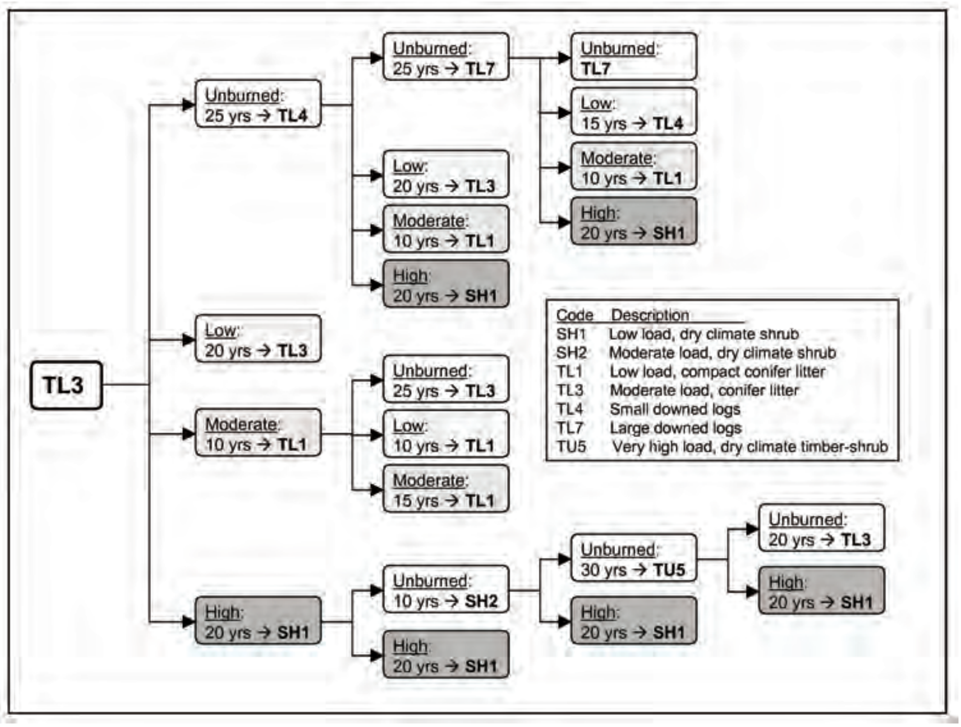

Figure 2 Sample diagram of dynamic fuel succession for the Timber-Litter 3 (TL3) fuel model. Transitions and their timing are described as a function of fire severity (low, moderate, high, or unburned).

height, canopy base height, and canopy bulk density. Weather data input to FARSITE included hourly wind speed and wind direction and daily minimum and maximum values for temperature and relative humidity. Outputs from FARSITE included fire perimeters at user-specified time intervals as well as fire behavior characteristics such as rate of spread, fireline intensity, and flame length. In addition to simulating surface fire spread, we used FARSITE to estimate crown fire activity for each of the modeled fires, which we then used as a proxy for burn severity.

We summarized historical weather data and computed fire danger indices with the analysis tool FireFamilyPlus (Bradshaw and McCormick 2000). We calculated daily percentile values for a fire danger index (energy release component, or ERC) and used these to inform our selection of ignitions and to identify days of active fire spread. Additionally, we used FireFamilyPlus to identify fire-ending events (i.e., precipitation exceeding a threshold within a certain time period) and to export the formatted weather and wind input data files required for FARSITE simulations.

Analysis We were interested in modeling suppressed ignitions during 1994–2004 that would have had the potential for significant spread. Even without suppression, many ignitions recorded in the fire occurrence database may never have spread from their ignition point due to fuel discontinuities, high fuel moistures, or subsequent weather conditions (e.g., rain). We used a combination of fire danger and fuel model to estimate each ignition's potential for spread, and ignored those with low potential. We identified two types of fuel models—"fast" and "slow"—in terms of a characteristic rate of fire spread. For example, we considered a fully cured tallgrass fuel model as "fast," whereas we considered a low-load conifer litter fuel model as "slow." Some fuel models were considered "slow" early in the fire season and "fast" later in the fire season after curing. If an ignition occurred in a "fast" fuel model, and the fire danger on the ignition date exceeded its 15th percentile value, we included the ignition in the retrospective analysis. If an ignition occurred in a "slow" fuel model, we included it if the fire danger exceeded the 50th percentile value. In a few cases, we subjectively relaxed these fire danger thresholds if an ignition didn't exceed the threshold but had other attributes indicating a potential for significant spread (e.g., when fire records indicated that the actual historic fire grew to >1 ha before containment).

The retrospective analyses included accounting for real fires that occurred during the study period as well as the simulation of the spread of the selected ignitions. For each year from 1994 through 2004, we constructed a timeline of fire danger values, ignition dates, significant weather events, and occurrence of real fires (wildfires, wildland fire use, and prescribed fires) that could have affected, or been affected by, the behavior of the fires we modeled. In chronological order, we used this information and the model FARSITE to simulate the spread and consequences of each of the selected ignitions. For example, if a real prescribed fire occurred in the study area before one of our selected ignitions occurred, we adjusted the fuels data to reflect the prescribed fire's effects before modeling the ignition. Other information in the timeline had the potential to further refine our ignition selection. For example, we eliminated an ignition from our analysis if fuels had not yet recovered from an earlier modeled fire that burned over the same location. Real fires were eliminated in the same fashion.

Fire spread and behavior were simulated by FARSITE using the actual weather and wind observations from the time period during which the fire would have burned. FARSITE tends to over-predict spread rates (Finney 1994) and these errors can accumulate over very long simulations. Furthermore, very long simulations can be computer intensive. To help mitigate these problems, we made two key simplifying assumptions. First, we assumed that the vast majority of fire spread occurs on those days with the most extreme fire weather conditions. Therefore, we only simulated fire spread on days when the fire danger exceeded the 90th percentile. We felt that using this threshold would capture the significant fire spread while balancing FARSITE's tendency to over-predict spread rate. Second, we assumed that fires would be extinguished if 0.5 inches of precipitation occurred within a three-day period, or if the end of the fire season was reached.

We determined burn severity from the fuel model and FARSITE's categorical estimate for crown fire activity. For non-timber fuel models such as grasses and shrubs, we assumed that fires always result in high fuel consumption, and, therefore, high burn severity. For timber fuel models, we determined burn severity from crown fire activity. Crown fire activity takes one of three values: surface fire, passive crown fire (torching), and active crown fire. We assumed surface fires would result in low severity, passive crown fires would result in moderate severity, and active crown fires would result in high severity. After each analysis year, we used the fuel succession model to update the fuels data according to the estimated burn severity within the simulated fire perimeters. We also updated fuels data for any real fires that may have burned using burn severity data that were available (Thode 2005). In cases where these data were unavailable, we assumed a uniform burn severity.

We summarized the cumulative effects of suppression by comparing the FRID map that would have existed at the end of the study period (2004) had our modeled fires been allowed to burn with the FRID map that existed without our modeled fires. To derive the FRID map with our modeled fires, we rebuilt the digital fire atlas using a GIS to incorporate the modeled fire perimeters, as well as any real fires that weren't eliminated by the modeled fires (Figure 3a). This rebuilt atlas was used in conjunction with fire return interval data to create the FRID map that would have existed in 2004 with our modeled fires. We used the same procedure, but using the original fire atlas with only the real fires that occurred, to derive the FRID map without the modeled fire (Figure 3b).

Results

Park records indicate that 34 lightning ignitions in the SFM watershed and 71 in the Kaweah watershed were suppressed during the period 1994–2004. Through ignition selection procedures, we identified 10 in the SFM watershed and 32 in the Kaweah watershed as having potential to spread significantly. Several of these were subsequently eliminated from our analyses because of effects from previously modeled fires. Ultimately, we modeled the spread of five ignitions in the SFM watershed and 23 in the Kaweah watershed. According to the model outputs, the five ignitions in the SFM watershed would have burned a total of 13,661 ha (43.5% of the watershed) and the 23 ignitions in the Kaweah watershed would have burned a total of 55,765 ha (61.5% of the watershed; Table 1).

Retrospective modeling indicates that the five ignitions from the SFM watershed and the 23 from Kaweah would have resulted in substantially lower values for FRID in 2004 compared with the FRID that resulted in their absence (Figure 4). For the SFM watershed, the average FRID would have decreased from 4.5 to 1.8, while in Kaweah it would have decreased from 4.3 to 0.3.

Discussion

The effects of the modeled fires on FRID were dramatic. Some of the modeled fires were much larger than what would ever be acceptable (Table 1). The simulation results suggest that the ignitions from 1994 and 1999 in the SFM watershed would have burned approximately 20% of the watershed in each year and would have escaped the park boundary. In the Kaweah watershed, the ignitions in 2001 would have burned almost a third of the watershed. Although fires of this size are not unprecedented (Caprio 1999), in reality, many of the modeled ignitions would have required management actions to confine them. We did not simulate confinement strategies in this study. A fruitful extension of this study would be to apply a more realistic "appropriate management response" scenario and examine the effect on FRID.

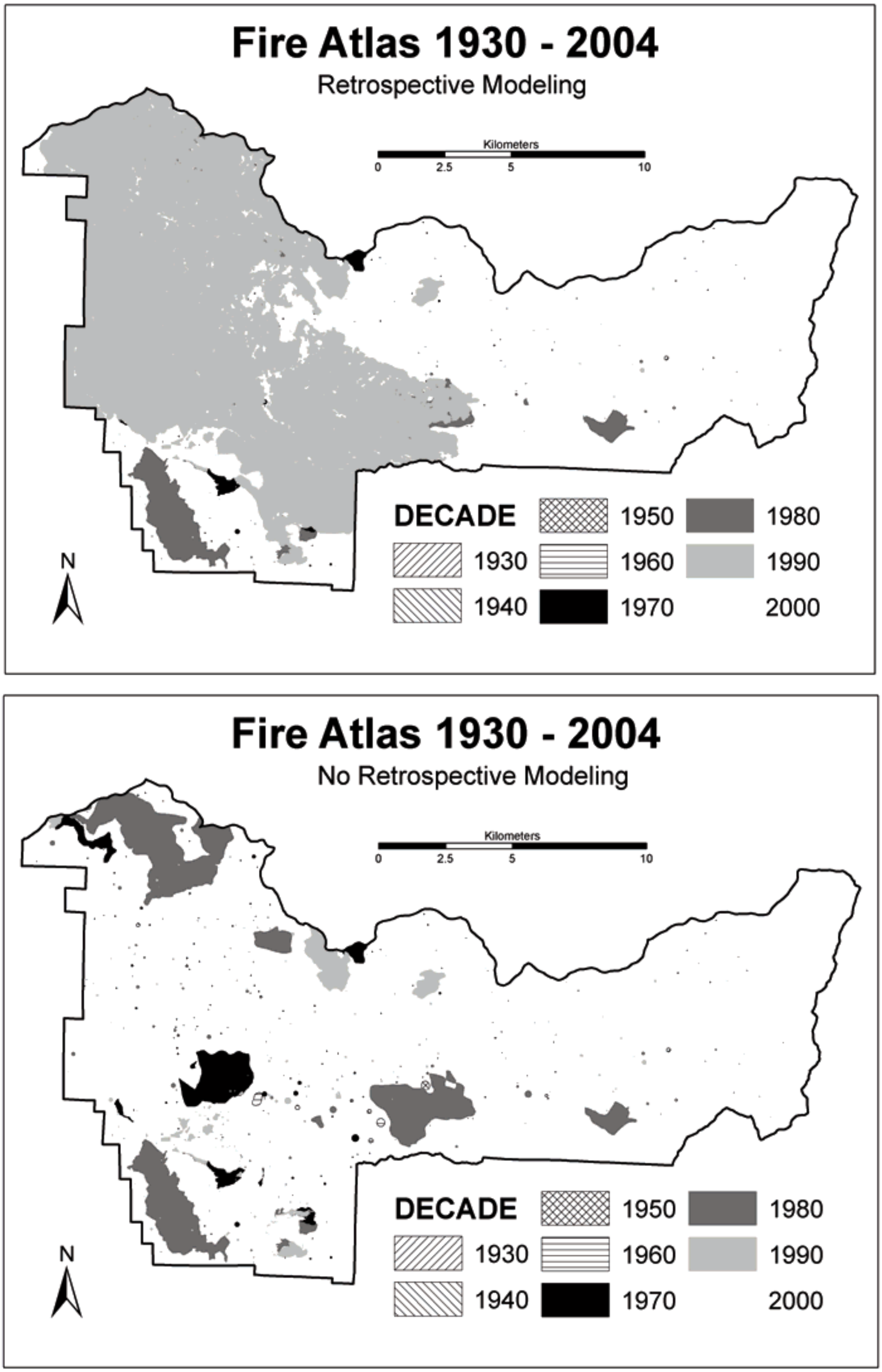

Figure 3 Digital fire atlases used to create the Fire Return Interval Departure (FRID) Index for the South Fork Merced (SFM) watershed. Figure 3a (above): Atlas built using the five simulated fire perimeters. Figure 3b (below): Atlas built without the simulated fires.

	South Fork Merced		Kaweah	
Year	*Number of ignitions simulated*	*Area burned (ha)*	*Number of ignitions simulated*	*Area burned (ha)*
1994	3	6,203	2	471
1995	0	0	4	3,120
1996	0	0	1	421
1997	0	0	0	0
1998	0	0	1	11
1999	1	6,809	1	1,336
2000	1	649	0	0
2001	0	0	3	28,862
2002	0	0	2	713
2003	0	0	6	9,071
2004	0	0	3	11,759
Total	**5**	**13,661**	**23**	**55,765**

Table 1 Area burned in retrospective simulations of suppressed lightning-caused ignitions in the two case study watersheds.

Our results hinge upon several assumptions, and at a minimum, sensitivity testing should be done for the simulation thresholds and fuel-model succession transition times we used. A lower ERC threshold for simulating days of active fire spread could dramatically increase the size of the modeled fires. The thresholds we used to select ignitions can also greatly affect the analysis because the removal or selection of any particular ignition can affect the selection or spread of subsequent ignitions. We recommend that information from initial size-up and scouting activities be used to improve the ignition selection. The transition times assumed in the fuel succession model may need refinement. If fuels recover more quickly than assumed, modeled fires could be even larger.

Both parks have fire management plans with extensive zones where the option of using natural ignitions to return fire to the landscape exists. Ideally, the decision to suppress a fire (or not) considers the possible consequences of allowing a fire to burn *as well as* the consequences of suppression. The analyses we conducted provide information about the consequences of suppression that could help inform decisions about future ignitions. Furthermore, knowledge of where fires would have burned naturally can help managers set priorities for fuel projects and, possibly, analyze opportunities for restoring “lost” ignitions with prescribed burns.

While parks and other protected areas strive to restore the natural role of fire, they must also protect a variety of other societal values such as air quality and public safety. Retrospective analyses can be applied to assess other consequences of suppression. The cumulative effects of suppression could be quantified in terms of smoke emissions over time, potential fire intensities, or even numbers of initial attack efforts that wouldn’t have been necessary if earlier ignitions had been allowed to burn. An understanding of what was gained and what

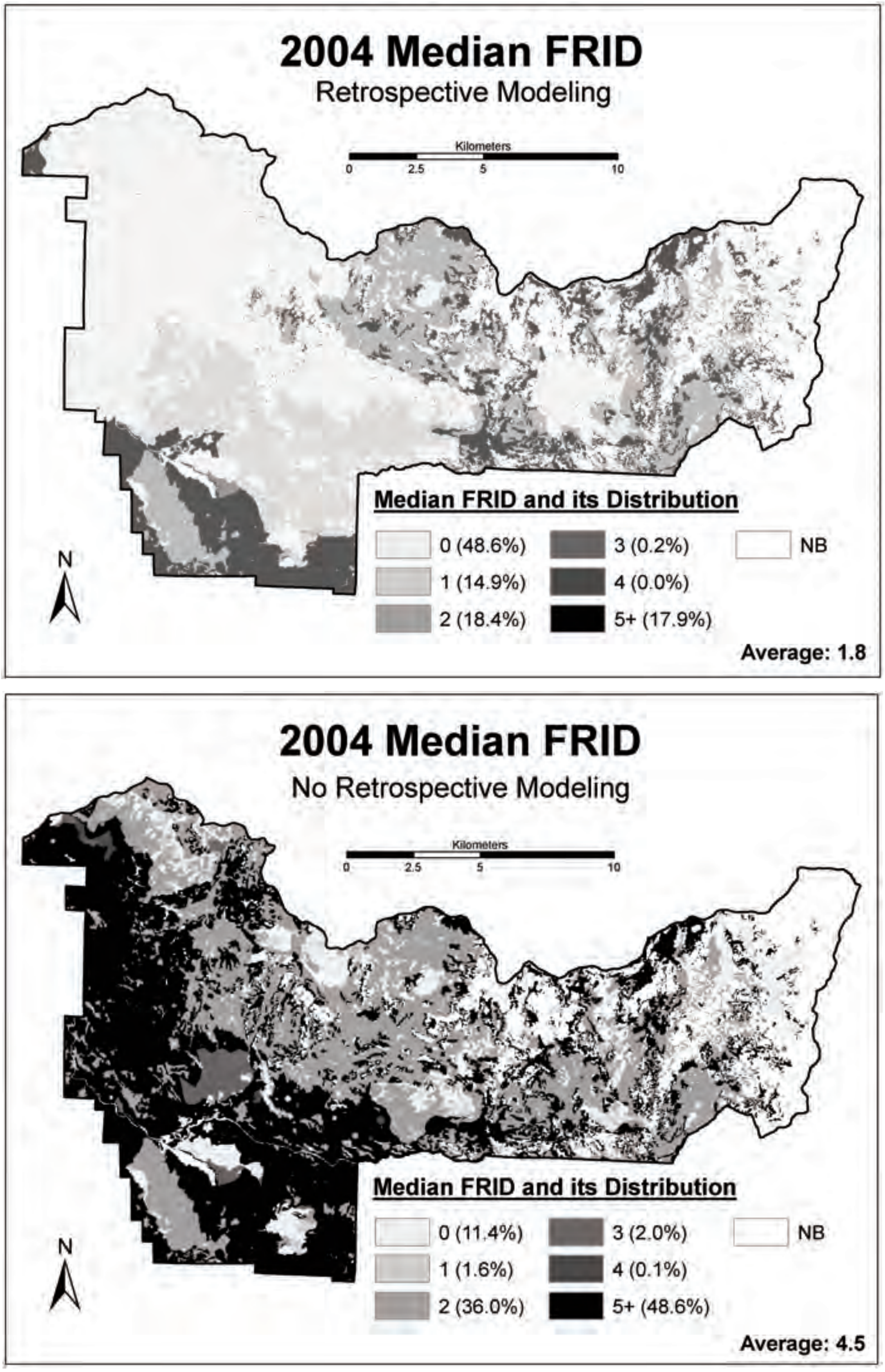

Figure 4 The Fire Return Interval Departure (FRID) Index for the South Fork Merced (SFM) watershed. Figure 4a (above): FRID derived with the five simulated fire perimeters. Figure 4b (below): FRID derived without the simulated fires.

was lost when each ignition was suppressed in the past is needed before managers can effectively communicate these tradeoffs to the affected public and neighboring governmental entities.

Conclusion

To accurately assess progress toward management objectives, park managers need an understanding of what was gained and what was lost when each ignition was suppressed in the past. When fires are suppressed, opportunities are foregone to create fuel breaks, reduce fire regime departures, and decrease future extreme fire behavior by modifying fuels. To our knowledge, no one has attempted to quantify these foregone opportunities. We developed a set of analysis steps to model suppressed ignitions in order to examine where these historic fires might have spread and to determine what effects they might have had on the landscape had they not been suppressed. This retrospective modeling approach is a quantitative method that park managers can use to better understand, measure, and track the cumulative effects of their decisions from year to year.

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