*Ecological Applications,* 10(1), 2000, pp. 85–94


# MODELING THE EFFECTS OF FIRE MANAGEMENT ALTERNATIVES ON SIERRA NEVADA MIXED-CONIFER FORESTS

CAROL MILLER[1,3] AND DEAN L. URBAN[2]

[1]*Graduate Degree Program in Ecology, Colorado State University, Fort Collins, Colorado 80523 USA*
[2]*Nicholas School of the Environment, Duke University, Durham, North Carolina 27708 USA*

*Abstract.* Decades of fire exclusion have led to hazardous fuel accumulations and the deterioration of fire-dependent ecosystems, particularly in the American West. Managers are striving to return the ecological role of fire to many ecosystems and would benefit from a much better understanding of how forest structure and composition might change if fires were reintroduced. We used a forest gap model, developed for forests in the Sierra Nevada, California, USA, that integrates climate, fire, and forest dynamics to investigate forest response to changes in the fire regime. The model simulates a spatially heterogeneous fuel bed that is responsive to changes in forest condition, making it well suited for examining alternative management approaches for restoring Sierra Nevada forests after a century of fire exclusion. Presuppression forest basal area, species composition, and spatial autocorrelation structure were restored quickly, if simulated disturbances that caused substantial tree mortality were reintroduced. Simulations of harvest induced the highest levels of mortality and, thus, most effectively restored forest structure and composition. However, prescribed fires were just as effective in restoring forest structure and composition if they were sufficiently severe.

*Key words: fire exclusion; fire reintroduction; fire suppression; forest gap model; fuel treatment; mixed-conifer forest; prescribed fires and forest restoration; Sierra Nevada, California (USA); spatial autocorrelation; structure vs. process debate.*

## INTRODUCTION

Decades of fire exclusion have profoundly changed many forests in North America, particularly in the American West. In the past, fires burned frequently in many forests, keeping fuel loads at a low level and maintaining relatively open forest stand structures (Kilgore 1987). These fires likely affected many other ecological processes by increasing rates of nutrient loss and cycling, altering the regeneration environment for many plant species, and altering soil structure and soil water holding capacity (Kilgore 1987). Fire exclusion during the past century has led to increased surface fuels, tree densities, and the continuity of aerial and surface fuels, thereby escalating fire hazard in many forests. The United States Forest Service estimates that 14–16 Mha of National Forest System land are at risk for catastrophic fire, with most of this area in the inland West (U.S. Congress 1998).

Land managers are seeking ways to reduce the threat of catastrophic fires and to restore the ecological role of fire in these fire-adapted ecosystems. The use of prescribed or natural wildland fires may accomplish these goals. In some cases, however, the hazard may be deemed too high and mechanical treatment of fuels may be required before fire can be safely reintroduced. For example, the Forest Service estimates that up to 90% of the 14–16 Mha that are at risk for catastrophic fire may require mechanical treatment before fire can be used as a tool for maintaining these fire-adapted ecosystems (U.S. Congress 1998).

In the Sierra Nevada of California, the effects of fire exclusion are most apparent in the low and middle elevation conifer forests. Before the 1850s, these forests experienced low- to moderate-severity surface fires (Wagener 1961, Kilgore and Taylor 1979, Swetnam 1993). During >80 yr of fire exclusion, shade tolerant species such as white fir have grown up in the understory, creating ladder fuels that could provide a path for fire to reach the crowns of canopy trees. Tree densities have increased, and there is an abundance of dead surface fuels in many of these forests (Parsons 1978, Vankat and Major 1978, Parsons and DeBenedetti 1979, van Wagtendonk 1985). Because of the dramatic effects of fire exclusion on these forests, and because of the preservation-oriented mission of the National Park Service, the national parks in the Sierra Nevada have a general goal of restoring presettlement forest conditions, with fire being used as a tool to attain this goal (Bancroft et al. 1985).

At the heart of this management goal of restoration is the structure vs. process debate. ''Structural restorationists'' insist that the forest structure needs to be restored to a state that would have existed had fires not been excluded, before letting natural fire maintain the

Manuscript received 13 May 1998; revised 15 December 1998; accepted 25 January 1999.
[3] Current adress: Aldo Leopold Wilderness Research Institute, U.S. Forest Service, Box 8089, Missoula, Montana 59807 USA. E-mail: cmiller/rmrs_missoula@fs.fed.us

ecosystem (Bonnicksen and Stone 1985). ''Process-based restorationists'' argue that the process of fire can restore the forest structure, by reducing live and dead fuels and opening up opportunities for fire-adapted species to grow (Parsons et al. 1986). A blending of these two views may be necessary for the restoration of these forests (Agee and Huff 1986). Stephenson (1996) suggests that process-based restoration without mechanical treatment can restore and sustain some aspects of pre-European sequoia grove structure, but recognizes that this approach may have its limitations. For example, some white fir trees that have established only because of fire exclusion may now be large enough to survive a fire, in which case, mechanical intervention may be warranted. Agee and Huff (1986) suggest that, for ecosystems with short fire return intervals, either structural goals or a hybrid of structural and process-based goals are appropriate, whereas for ecosystems with longer fire return intervals, process-based goals are usually appropriate.

We developed a version of a forest gap model for Sierra Nevada forests that integrates climate, fire and forest dynamics (Miller and Urban 1999*a*). The model simulates spatially heterogeneous fuels that are responsive to changes in forest condition, making it suitable for investigating structure- and process-based approaches for restoring these forests after a century without fire. Our objective was to demonstrate how Sierra Nevada forests might respond to different management strategies for reintroducing fire. We used the model to examine forest response to three hypothetical strategies representing differing degrees of disturbance severity. To assess forest stand dynamics, we examined changes in average basal area and species composition. To examine the explicit spatial pattern within the simulated stands, we computed the spatial autocorrelation of basal area.

## Methods

We extended a forest gap model (Smith and Urban 1988, Urban et al. 1991) for the Sierra Nevada by adding a new soil moisture model (Urban et al., *in press*), a new fire model (Miller and Urban 1999*a*, *b*), and parameterizing it for Sierran mixed-conifer forests. Here we provide specific details necessary for understanding the results presented in this paper. For further detail, the reader is referred to Miller and Urban (1999*a*, *b*).

The model simulates a forest stand as a rectangular grid of small ''tree-sized'' (15 × 15 m) cells; cells interact with each other via the light regime, where tall trees on a cell cast shade on neighboring cells (Urban et al. 1991). The modeled grid has a user-specified elevation, slope, and aspect, thus representing a slope ''facet'' (Daly et al. 1994). Climate parameters are internally adjusted for elevation and topography (Urban et al., *in press*). In this paper, we use a grid of 20 × 20 cells to simulate a 9-ha forest stand. We refer to this version as the FACET Model, or simply FM; the benchmark edition described here is version FM 97.5.

Like other forest gap models, FM simulates seedling establishment, annual diameter growth, and mortality for individual trees on a small model plot (a grid cell in FM). Each of these demographic processes is specified as a maximum potential that can be achieved under optimal conditions. These potentials are then reduced to reflect suboptimal environmental conditions (e.g., low light or drought) on each cell. A key characteristic of gap models is that they simulate system feedbacks: not only are trees affected by their environment, but each tree exerts an influence on its environment (e.g., through shading and transpirational demand).

The environmental factors that can limit tree growth in FM are available light, soil moisture, temperature, and nutrient availability. Available light is estimated for each position within the stand as a function of the leaf area, which is distributed vertically along each tree's crown (Urban et al. 1991). Soil moisture is estimated from precipitation, potential evapotranspiration, and the water holding capacity of the soil, from which a drought-day index is computed (Miller and Urban 1999*b*; Urban et al., *in press*). Growing degree-days are computed and used as an index for temperature. Finally, nutrient availability is estimated as a ratio of nitrogen uptake and nitrogen made available through decomposition of forest litter (Miller and Urban 1999*a*; Urban et al., *in press*). The available light, number of drought-days, growing degree-days, and nutrient availability on each grid cell define the environment for each tree in that cell.

Species tolerances to shade, drought, and temperature govern each tree's growth response to the environment in each grid cell (Miller and Urban 1999*a*; Urban et al., *in press*). Differential species response to nutrient availability has been turned off in this version of the model to minimize the model's sensitivity to this uncertain parameter. Nine tree species are simulated in this version of the model: white fir (*Abies concolor* [Gord. and Glend.] Lindl. ex Hildebr.), red fir (*Abies magnifica* A. Murr.), incense cedar (*Calocedrus decurrens* [Torr.] Floren), lodgepole pine (*Pinus contorta* Dougl. ssp. *murryana* Grev. and Balf.), Jeffrey pine (*Pinus jeffreyi* Grev. and Balf.), sugar pine (*Pinus lambertiana* Dougl.), western white pine (*Pinus monticola* Dougl.), ponderosa pine (*Pinus ponderosa* Laws.), and California black oak (*Quercus kelloggii* Newb.).

### *The fire model*

This version of FM was developed to study surface fire regimes; thus, the model considers only ''dead and down'' woody fuels and litter. Because live vegetation is not included in the fuel bed, this model neither simulates crown fires, nor the transition from surface fires to crown fires. Fuels accumulate as a function of site environment and forest conditions. Each year during a simulation, a fraction of each tree's foliage and branch wood is added to the fuel bed according to species-specific allometries (Miller and Urban 1999*a*). In addition, biomass from dead trees is gradually added to the fuel bed. Woody fuels and litter are classified by size using the conventions of fire behavior and fire danger models (Deeming et al. 1972). Each fuel class decays according to a constant rate that is modified by an abiotic decay multiplier describing the temperature and moisture environment of the site. Decay rates for each fuel class were calibrated to data from Sequoia–Kings Canyon and Yosemite National Parks (Parsons 1978; J. van Wagtendonk, *unpublished data*). Herbaceous fuels are also simulated in this version of the model, but their importance is limited to elevations <1500 m (Miller and Urban 1999*b*; Miller and Urban, *in press*).

Because FM is implemented as a grid of small ''tree-sized'' cells, it can describe the spatial heterogeneity of forest structure and composition that exists within a stand. Fuel inputs, and therefore fuel bed conditions, vary temporally and spatially throughout a stand according to the number, size, and species of trees that are present. In addition, the fuel moisture varies both temporally and spatially with the local site water balance. Fuel moisture is estimated from the soil moisture content of the forest floor layer, computed in FM's soil moisture model (Urban et al., *in press*). Thus, as FM generates spatial heterogeneity in forest structure and condition due to tree-level processes, this leads to heterogeneity in fuel bed conditions, thereby generating spatial patterns in fire intensity and effects.

We consider FM to simulate a ''natural'' fire regime, as both fire frequency and magnitude (i.e., area burned) are generated internally by the model and are governed by site conditions. Natural-fire events are simulated as a function of three factors: probability of ignition, fuel load, and fuel moisture. The mean ignition interval, in years, for the model grid is specified at run time, and uniform-random numbers are drawn to generate stochastic ignition events around this mean interval. In this paper, we assume that ignitions are not limiting, and we set this interval so that an ignition occurs every year. However, for a fire to occur from an ignition, low fuel moisture and sufficient fuel loadings must also exist. Because the soil–water balance (and thus fuel moisture) becomes more positive with elevation, FM generates a decreasing fire frequency with elevation; the simulated pattern agrees well with independent fire history data for the study area (Miller and Urban 1999*a*).

When an ignition occurs, the fire-line intensity is computed for each of the grid cells from the accumulated fuels and fuel moisture conditions, following well-established equations for surface fire behavior (Rothermel 1972, Albini 1976). Only cells with computed intensities >45 kW/m are considered to be burnable. This intensity is roughly equivalent to a scorch height of about 0.5 m, and we assume that fires ''burn out'' when intensities are less than this. Fires may spread to all cells within the model grid, but they are restricted to those cells that are burnable and that are also spatially contiguous to a randomly located ignition point on the grid. Thus, fires are restricted to a contagious cluster of burnable cells, and, on average, fires tend to burn the largest cluster of burnable cells. Although this approach does not simulate the complexities of fire spread, FM successfully reproduces empirical relationships between area burned and fire frequency that have been inferred for presettlement fire regimes (Miller and Urban 1999*a*).

Fire effects are calculated for each cell that burns. Fuels are reduced as a function of prefire fuel load (Brown et al. 1985), scorch height is calculated as a function of mean daytime temperature and fire-line intensity (Van Wagner 1973), and fire-induced tree mortality is computed as a species-specific function of crown scorch (Ryan and Reinhardt 1988, Stephens 1995, Mutch and Parsons 1998).

In addition to a natural-fire regime, three new disturbance options are available in this version of the model: prescribed fire, harvest, and suppression. In the prescribed-fire option, the timing of prescribed fires is specified at run time. When a prescribed fire occurs, every cell within the simulated stand burns. Fire-line intensity is rescaled so that the range of intensities is 45–345 kW/m, a range consistent with management prescriptions in mixed conifer forests (van Wagtendonk 1974). Thus, prescribed fires in FM differ from the simulated natural fires in two ways. First, fire intensity for a prescribed fire is limited at its upper range, whereas natural fire has no upper limit. Second, the area burned by prescribed fires is always the size of the entire model grid, whereas natural fire might burn only a subset of the model grid. In the harvest option, the year in which cutting occurs and the maximum tree diameter cut are specified at run time. All trees up to the maximum diameter are removed from the site. All bole wood from these is removed, but 30% of the foliage and branch wood is left on site as a conservative estimate of collateral residue from harvesting activities. In this paper, we use a maximum diameter of 35 cm for our harvest. The start and end year of a suppression era can be specified at run time, as well. Prescribed fires and harvest operations may occur during suppression, but no natural fires are simulated during a suppression era.

### *Simulations*

We simulated three different disturbance scenarios for south-facing 20% slopes at 1850, 2050, and 2250 m elevation. The first 800 yr of each disturbance scenario were identical: we ran the model from bare ground for the first 200 yr without fire, followed by 500 yr of FM's internally generated natural-fire regime (ignition interval = 1), and finally by 100 yr without fire. The first 200 yr were required to allow succes-

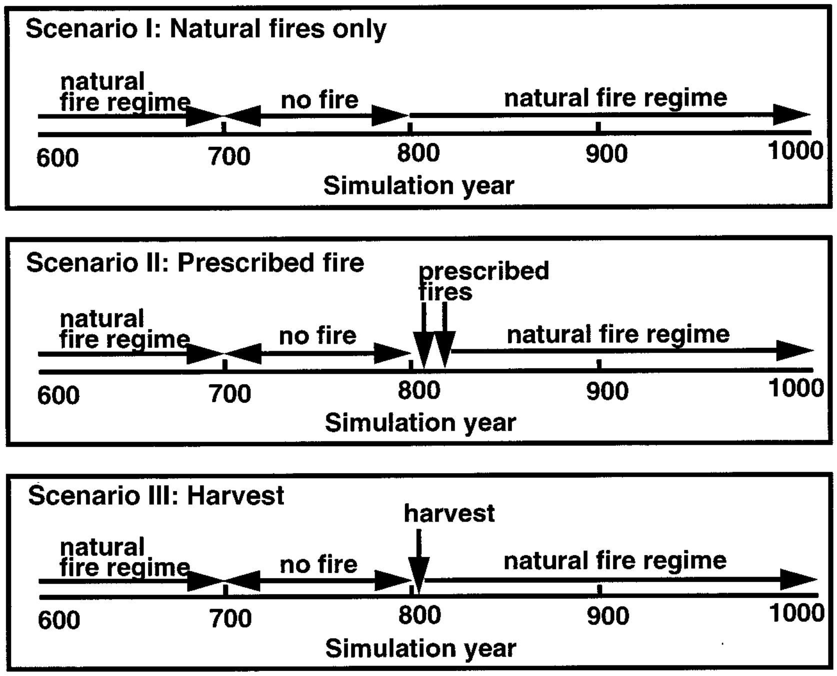


FIG. 1. Three disturbance scenarios. In Scenario I, natural fires occur after simulation year 800. In Scenario II, prescribed fires in simulation years 801 and 811 precede natural fires. In Scenario III, the removal of trees <35 cm dbh precedes natural fires.

sional trends and fuel bed bulk density to stabilize. We chose 500 yr of a natural-fire regime to ensure that stand-level measurements of forest conditions had equilibrated to the fire regime. This portion of each simulation estimates the presettlement fire regime and forest condition. The subsequent 100 yr without fire approximates the current fire-free interval for most sites on the west slope of the Sierra Nevada in Sequoia National Park (Caprio and Swetnam 1995).

The three scenarios varied starting in simulation year 801 (Fig. 1). In the first scenario, we simulated the reintroduction of a natural-fire regime simply by allowing FM to generate a natural-fire regime after year 800. In the second scenario, simulated prescribed fires in years 801 and 811 preceded the reintroduction of a natural-fire regime. In the third scenario, the harvest of all trees <35 cm dbh in year 801 preceded the reintroduction of a natural-fire regime. These scenarios represent different levels of disturbance severity before the reintroduction of a natural-fire regime. All simulations were for Sequoia National Park (36.6° N, 118.6° W), and all simulations used a homogeneous sandy loam with soil depth of 1 m.

### *Analysis*

We examined how forest structure varied during the simulations as the disturbance regime changed. To assess changes in forest structure, we plotted total basal area against time. We also examined "large tree basal area" as the percent of the total basal area contributed by trees >60 cm dbh. Trees larger than this are considered either "mature" or "old" in field studies in the Parks (e.g., van Wagtendonk et al. 1996) and typically occupy codominant or dominant positions in the canopy. These size classes dominated the presettlement forest, which was kept open and park-like from frequent surface fires (Kilgore 1973). The development of a dense understory during fire suppression is reflected by a decrease in the percent basal area from large trees.

We also examined how species composition varied during the simulations as the disturbance regime changed. To examine changes in species composition through time, we estimated percent similarity (Mueller-Dombois and Ellenberg 1974) between the stand in the current simulation year and the mean stand condition during the last 100 yr of the presuppression era (simulation years 601–700). Percent similarity (PS) was calculated as follows:

$$PS = \frac{2M_w}{M_a + M_b} \times 100 \quad (1)$$

In our case, $M_a$ is the mean total basal area taken over simulation years 601–700, and $M_b$ is the total basal area in the current simulation year. $M_w$ represents the species composition that is common to the two stands being compared, and it is derived by summing the smaller of the two values of species basal areas for species that are common to both stands (Mueller-Dombois and Ellenberg 1974). A value of PS = 100 indicates that species composition in the current simulation year is identical to the mean presettlement forest, whereas PS = 0 indicates that none of the species from the presettlement forest are present. We computed percent similarity each year during the simulation.

We were also interested in the explicit spatial pattern within the simulated stands. From maps of basal area output at 20-yr intervals during the simulations, we calculated the spatial autocorrelation of basal area within the simulated forest stands. A variable is said to be spatially autocorrelated when it is possible to predict the values of this variable at some points in space from the known values at other sampling points. When autocorrelation is significant at a particular distance, it may indicate the scale of the underlying spatial pattern. We calculated an autocorrelation coefficient, Moran's *I* (Moran 1950), to detect spatial autocorrelation in the simulated forests. We computed Moran's *I* for the first 10 distance classes on the cell basal area at 20-yr intervals in the simulation output. We evaluated the mean of these results for 10 replicates of the simulations at 2050 m. Replicates were identical in all respects, except that each one used a different random number seed to generate the pseudorandom weather used by the model. The same random number seeds were used across each of the three scenarios.

## Results

### *Structural patterns*

Total basal area increased in the absence of fire during years 701–800, as the stands became more dense and individual trees grew larger (Fig. 2). Disturbance was reintroduced after year 800 and total basal area declined in all scenarios. The most dramatic effect occurred in the harvest scenario, which removed all trees <35 cm dbh and, thus, removed a significant amount of basal area. In the natural-fire scenario, total basal area gradually declined and reached its presuppression levels well before the end of the simulation. The forest response in this scenario was gradual, taking 150–400 yr, because the simulated natural fires were of relatively low severity. The major effect of natural fires was to kill small trees and saplings, thus maintaining a sparse understory. Most of the basal area reduction in the natural-fire scenario occurred as trees that established during fire suppression eventually died from either competitive interactions or age-related mortality.

Before the fire suppression era, the percentage of basal area due to large trees (>60 cm dbh) was 70–80% (Fig. 3). Frequent low intensity fires before year 700 killed many understory trees but left large trees unharmed, skewing the distribution of trees toward larger size classes. During suppression, from year 701–800, the percentage of the total basal area contributed by large trees declined as new trees established in the understory and contributed significantly to the total basal area. After year 800, when disturbance was reintroduced in the simulations, the proportion of basal area

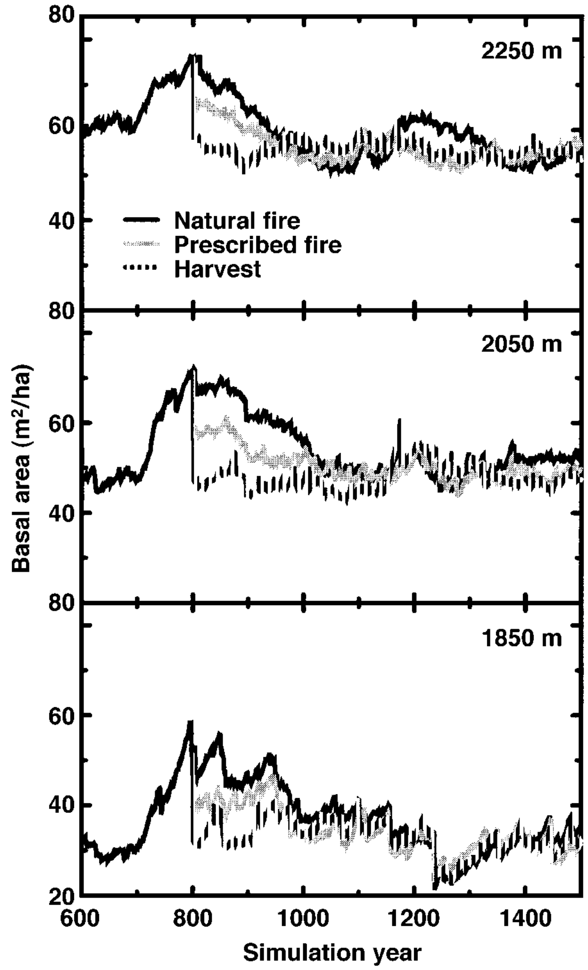


FIG. 2. Response of total basal area to changes in the disturbance regime for three sites.

from large trees increased again as understory trees were removed. Relative basal area of large trees increased most rapidly in the harvest scenario; at its peak in year 900, it was much higher than the other two scenarios. This rapid increase was due initially to the removal of trees <35 cm dbh in the harvest operation in year 801. The increase continued for another 100 yr as trees with diameters 35–60 cm grew into the large-tree (>60 cm dbh) size class. In all scenarios, the percentage of basal area from large trees returned to presuppression levels of 70–80% by the end of the simulation.

### *Compositional patterns*

Fire exclusion can result in shifts in species composition. During fire suppression, the stand became less similar to the reference condition (Fig. 4). At 2050 m and 2250 m, this change in similarity is primarily due to the increase in white fir basal area, a shade-tolerant and fire-sensitive species (Fig. 5). At 1850 m, the decrease in similarity is due to the increase of incense

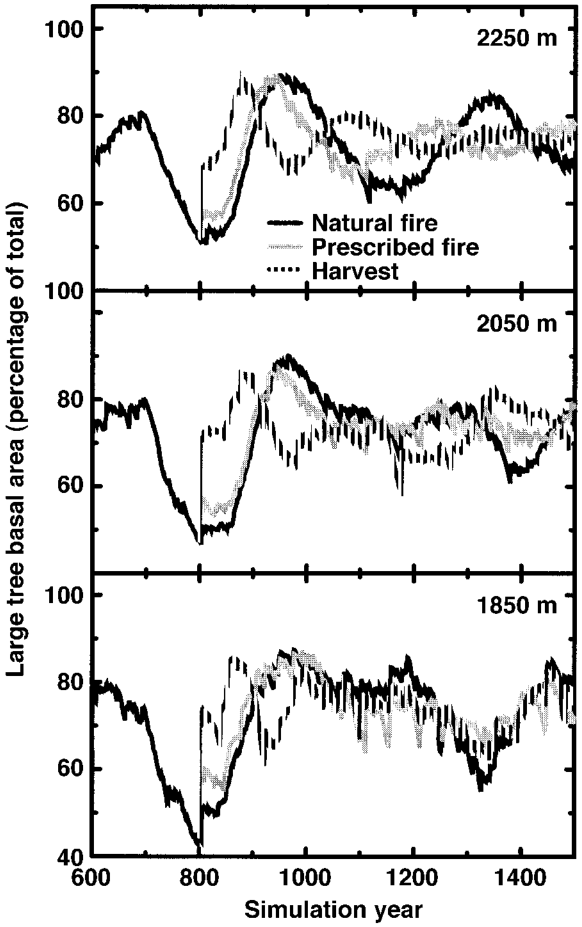


FIG. 3. Response of large tree basal area to changes in the disturbance regime for three sites. Large tree basal area is the percentage of total basal area contributed by trees >60 cm dbh.

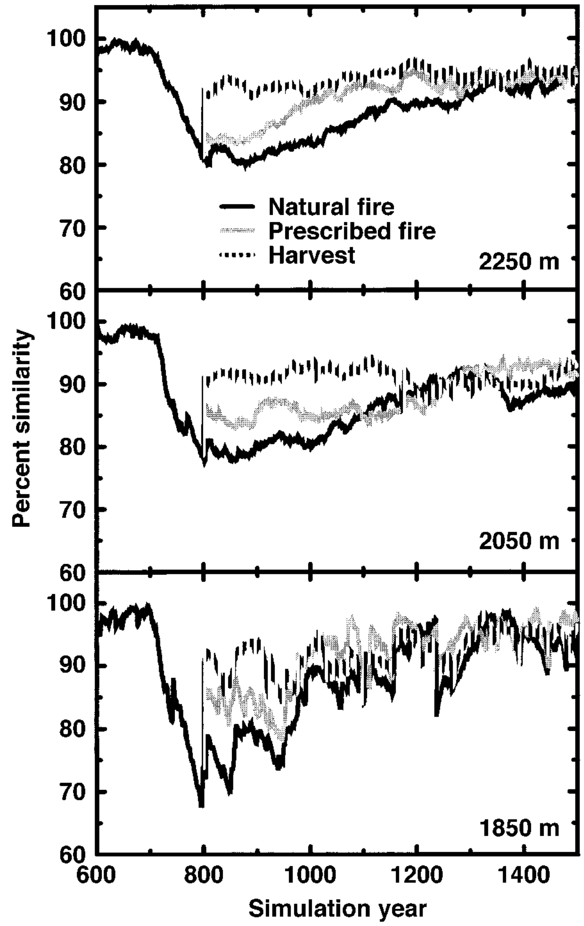


FIG. 4. Response of species composition to changes in the disturbance regime for three sites. Percent similarity is measured relative to the mean stand condition, with means evaluated over simulation years 601–700 (the last 100 yr of the presuppression era).

cedar basal area (Fig. 5). When disturbances were reintroduced after year 800, the stands became more similar to the reference condition again, as the fire-sensitive trees are removed (Fig. 4 and 5). The increase in percent similarity after year 800 was most pronounced in the harvest and prescribed-fire scenarios.

### *Spatial structure*

Although we computed Moran's *I* for the stand over the first 10 distance classes, it was significant only for the first distance class. Therefore, we have shown Moran's *I* plotted at 20-yr intervals only for the first distance class (Fig. 6). The lower 95% confidence limit is shown as a dotted line; points below this line represent significant negative spatial autocorrelation for distance class 1. Before suppression (years 600–700), there was no significant spatial autocorrelation. During suppression, however, basal area became significantly and negatively autocorrelated; by year 800, Moran's $I = -0.13$. When disturbance was reintroduced after year 800, the spatial autocorrelation became less significant with time. The prescribed-fire and harvest scenarios produced the most rapid responses, with a return to nonsignificant values before year 900.

## DISCUSSION

Mixed-conifer forests in the Sierra Nevada have experienced a dramatic change in their disturbance regime during this century (Swetnam 1993, McKelvey et al. 1996, Skinner and Chang 1996). Whereas frequent low-intensity surface fires burned through these forests for thousands of years in the past, the past century has been virtually fire-free. Changes in vegetation and fuel loads have been observed and inferred from a variety of sources (Vankat and Major 1978, Parsons and DeBenedetti 1979, van Wagtendonk 1985).

The harvest scenario induced the highest mortality and quickly restored at least three aspects of forest structure and composition to presuppression conditions. In the scenario with harvest, rapid changes occurred to total basal area (Fig. 2), large tree basal area percentage (Fig. 3), percent similarity of forest com-

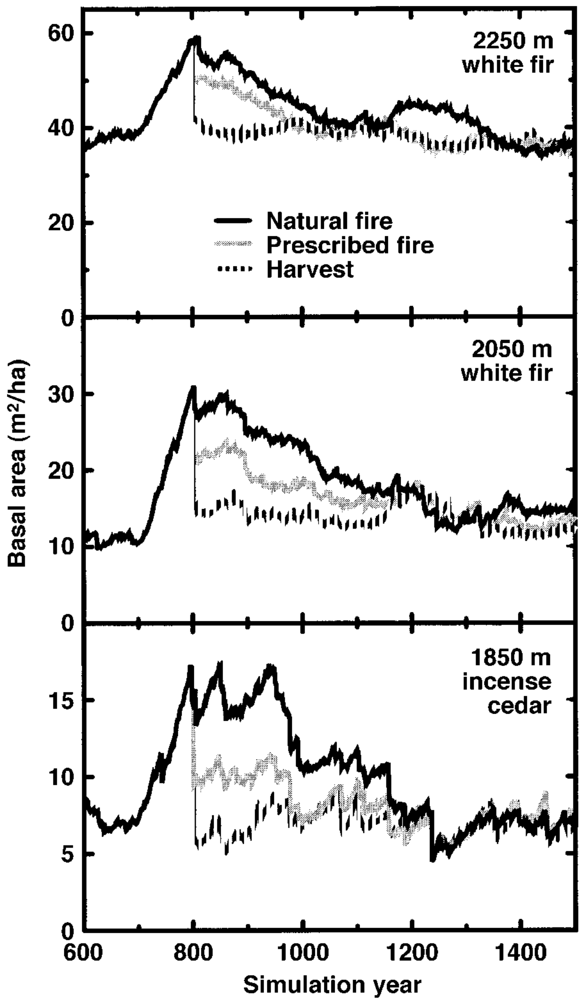


FIG. 5. Basal area response of the primary shade-tolerant, fire-resistant species to changes in the disturbance regime for three sites.

position (Fig. 4), and spatial autocorrelation of basal area (Fig. 6). Subsequent fires maintained these new structural conditions. The prescribed-fire scenario resulted in similarly rapid changes, although to a lesser degree, and the natural-fire scenario resulted in much slower and subtler adjustments.

The range of fire intensity (45–345 kW/m) assumed for prescribed fires does not result in as much mortality as the scenario where all trees <35 cm dbh are harvested. These intensities are not high enough to kill trees significantly >25 cm dbh. However, if prescribed fires burn with higher intensities, results similar to the harvest scenario can be achieved. Fig. 7a demonstrates the effect of increasing the fire intensity of prescribed fires. When prescribed fire intensity is within 345–1730 kW/m, the resulting basal area reduction is similar to that of the 35 cm dbh harvest scenario. For comparison, the effect of varying the maximum size of harvested trees in the harvest scenario is also shown (Fig. 7b).

In reality, natural fires may be more effective at restoring forest structure and composition than these results suggest. In the model, natural fires can be severe enough to kill large trees and substantially influence forest structure. The high degree of heterogeneity in fuel moisture and fuel loads generated by the model, however, keeps these areas of high severity localized to one or a few model cells. In the natural-fire simulation at 2050 m for example, average fire intensity after suppression was only 180 kW/m, but local intensities occasionally exceeded 8000 kW/m and resulted in 100% crown scorch. Although these locally intense fires have severe impacts on individual cells, they do not significantly reduce basal area at the stand level. The simulated prescribed fires, on the other hand, result in much less heterogeneity, because the model limits prescribed fire intensity to a narrower range (45–345 kW/m) and ensures the burning of the entire grid. In addition, the model may underestimate the effects of natural fires, especially after a period of fire suppression, because live canopy fuels are not simulated. These

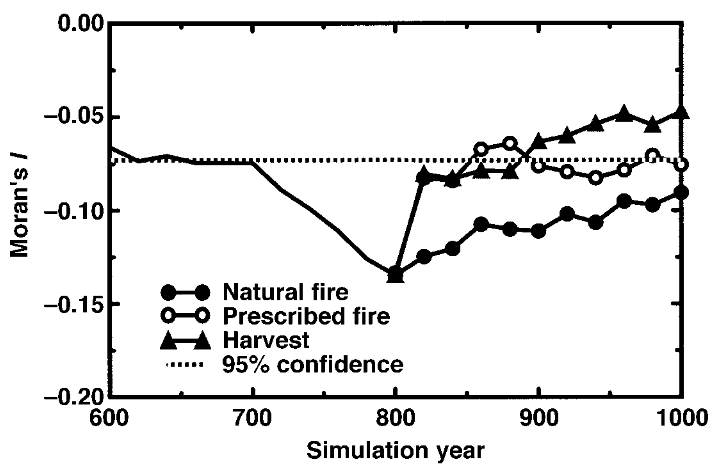


FIG. 6. Response of spatial autocorrelation of basal area to changes in the disturbance regime for the site at 2050 m elevation. Values shown are for distance class 1 and are the means of 10 replicate simulations. The lower 95% confidence limit is shown as a dotted line; points below this line represent significant negative spatial autocorrelation for distance class 1.

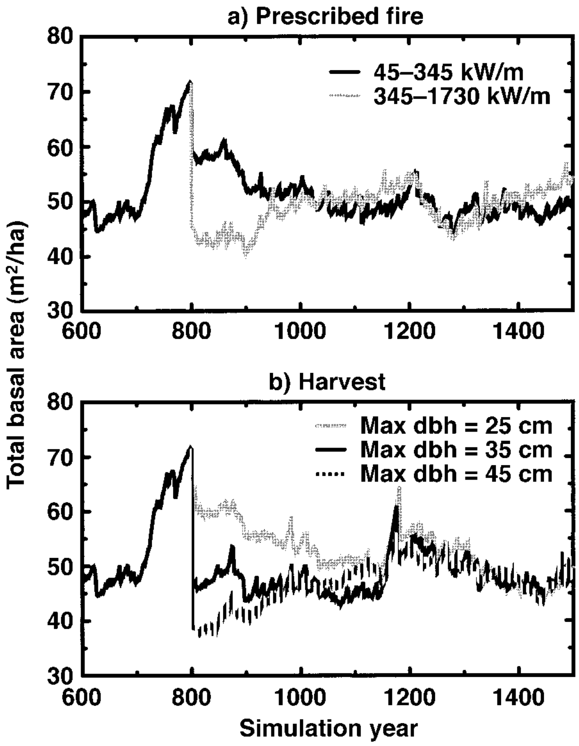


FIG. 7. Response of basal area to changes in the disturbance regime for the site at 2050 m elevation. (a) The prescribed-fire scenario using different ranges of prescribed fire intensity, and (b) the harvest scenario using different values for maximum dbh harvested.

fuels can serve as ladders and escalate fire hazard. Furthermore, because it does not simulate short-term weather fluctuations, the model may underestimate the extent and intensity of some fires. For example, smoldering fires with intensity <45 kW/m are assumed to burn out in this model. In a real fire, these smoldering fires can flare up into more intense fires when the weather changes and extreme conditions of wind, temperature, and relative humidity set in.

An important, yet relatively unexplored, aspect of forest structure is the spatial pattern within a forest. Forests are mosaics of groups of trees that originate in forest gaps caused by a variety of disturbances, including fire (Watt 1947, Bormann and Likens 1979, Pickett and White 1985, Stephenson et al. 1991). When we simulate fire suppression, gap formation is limited to the death of individual canopy trees and the spatial pattern in basal area that is created is a result of the light regime. Tall trees on one cell cast shade on neighboring cells, restricting the amount of basal area the neighboring cells can accumulate. The pattern is fairly regular at short lag distances. Thus, significant negative spatial autocorrelation develops at short lag distance classes (Miller and Urban 1999*b*). When larger disturbances are reintroduced, they obscure the neighborhood effects from the light environment, and spatial autocorrelation becomes nonsignificant. The transition to nonsignificant values of Moran's *I* occurs quickly after harvest or prescribed fires, but takes >200 yr in the natural-fire scenario (Fig. 6).

Baker (1995) simulated crown fire regimes and examined how long it took for landscape structure to adjust to a new fire regime. He found that most landscape indices equilibrated within one or two rotations of the new fire regime. Fire rotation time is the time required to disturb an entire landscape and is a function of both the frequency and mean size of fires. In our simulations, we found that spatial structure responded quickly to reintroduced disturbances, but only if the disturbances were severe enough to significantly affect basal area. Prescribed fires and the harvest operation resulted in a rapid return to presuppression levels. In the natural-fire scenario however, the adjustment takes much longer. From the mean fire size and mean fire interval, we calculated the mean fire rotation time for the natural-fire scenario as ~14 yr. Thus, when fires are not very severe, many more than two rotations of the new fire regime may be necessary for the measure of spatial autocorrelation to equilibrate.

These results suggest that restoration of presettlement forest conditions is accomplished most expediently when the reintroduced disturbance results in significant tree mortality. However, our results also suggest that restoration after fire suppression is possible when the initial disturbance is less severe (e.g., our natural-fire scenario). Natural fires that are reintroduced after suppression alter the establishment environment for seedlings, kill small saplings, and maintain a less dense understory. In time, forest structure returns to presettlement conditions, because trees that established only because of fire suppression and have become large enough to resist surface fires eventually age and die. Centuries are required for this method of restoration, which is a much longer planning horizon than managers currently use.

Because this version of the model does not simulate live canopy fuels that can serve as ladders and escalate fire hazard, we cannot yet address whether forest structure must be restored to presettlement conditions before natural fire can be safely used as a restoration tool. With improvements to the model that include ladder fuels, we may be able to assess this question in the future. In forests that historically have been dependent on surface fire regimes, fire suppression may result in a forest condition that increases the likelihood for catastrophic wildfires, and may even result in stand-replacing crown fires. In such cases, managers should intervene either mechanically or with prescribed fire to restore forest condition and reduce fuels before allowing naturally ignited fires to burn. Otherwise, our results suggest that natural fires can be used without other fuel manipulations to restore (albeit slowly) forest structure. The most effective natural fires will be those that result in substantial tree mortality. Managers who wish to use natural fire as a restoration tool have the difficult task of allowing substantial fire related tree mortality while preventing the catastrophic events that can threaten human life and property. Under certain circumstances (e.g., in many wildland–urban interface zones), unacceptable hazards to life and property may entirely preclude the use of fire as a tool for forest restoration.

When interpreting the results from any modeling exercise, one must do so with an understanding of the assumptions in the model. We have already mentioned assumptions in the simulation of fire behavior that may affect our interpretation of model results. Likewise, the model does not simulate several side effects of harvest operations. For example, the newly opened canopy after harvest can result in increased wind speeds and increased drying of fuels. In addition, soil compaction, erosion, and physical damage to standing trees could alter establishment success and mortality of trees. Thus, harvest may actually increase fire hazard (van Wagtendonk 1996).

Fire suppression may not be solely responsible for the long fire-free intervals in Sierran forests observed during this century. Intensive livestock grazing at the turn of century removed fine fuels, restricting fire spread and reducing fire frequency (Vankat 1977, Kilgore and Taylor 1979, Swetnam 1993). Native Americans burned many areas in the Sierra Nevada; the extirpation of native American populations also may have contributed to the decrease in fire frequency during certain time periods and particular locations (Vale 1998). The restoration of presettlement fire regimes may not be possible in areas where either intensive grazing continues or burning by native Americans was important in the past.

## Conclusion

Long periods of fire exclusion result in altered forest structure and composition of mixed-conifer forests in the Sierra Nevada. These simulations suggest that presettlement forest structure, composition, and spatial autocorrelation structure can be restored quickly, if disturbances are reintroduced that cause significant tree mortality. In the present simulations, an initial harvest induced the highest level of mortality, most effectively restoring forest structure and composition. Prescribed fires can be just as effective in restoring forest structure and composition, if they are sufficiently severe. It is not clear, however, that forest structure needs to be restored before natural fires are reintroduced. Extensions to this model are necessary before this question can be adequately addressed.

## Acknowledgments

Funding for this research was provided by the U.S. Geological Survey/Biological Resources Division Sierra Nevada Global Change Research Program under contract CA8800-1-9004. This research relied heavily on the collaboration and exchange of ideas among all scientists involved with the Sierra Nevada Global Change Research Program, including Dave Parsons, Nate Stephenson, Tom Swetnam, and Jan van Wagtendonk. This work was completed as part of Carol Miller's doctoral degree at Colorado State University. Thoughtful comments by Tom Swetnam and two anonymous reviewers greatly improved the manuscript.

## Literature Cited

Agee, J. K., and M. H. Huff. 1986. Structure and process goals for vegetation in wilderness areas. Pages 17–25 *in* Proceedings of the National Wilderness Research Conference: current research. U.S. Forest Service General Technical Report **INT-212**.

Albini, F. A. 1976. Estimating wildfire behavior and effects. U.S. Forest Service General Technical Report **INT-30**.

Baker, W. L. 1995. Long-term response of disturbance landscapes to human intervention and global change. Landscape Ecology **10**:143–159.

Bancroft, L., T. Nichols, D. Parsons, D. Graber, B. Evison, and J. van Wagtendonk. 1985. Evolution of the natural fire management program at Sequoia and Kings Canyon National Parks. Pages 174–180 *in* J. E. Lotan, B. M. Kilgore, W. C. Fischer, and R. W. Mutch, *technical coordinators.* Proceedings of the symposium and workshop on wilderness fire. U.S. Forest Service General Technical Report **INT-182**.

Bonnicksen, T. M. and E. C. Stone. 1985. Restoring naturalness to national parks. Environmental Management **9**: 479–486.

Bormann, F. H., and G. E. Likens. 1979. Pattern and process in a forested ecosystem. Springer–Verlag, New York, New York, USA.

Brown, J. K., M. A. Marsden, K. C. Ryan, and E. D. Reinhardt. 1985. Predicting duff and woody fuel consumed by prescribed fire in the northern Rocky Mountains. U.S. Forest Service Research Paper **INT-337**.

Caprio, A. C, and T. W. Swetnam. 1995. Historic fire regimes along an elevational gradient on the west slope of the Sierra Nevada, California. Pages 173–179 *in* J. K. Brown, R. W. Mutch, C. W. Spoon, and R. W. Wakimoto, technical coordinators. Proceedings of the symposium on fire in wilderness and park management, Missoula, Montana, 30 March–1 April 1993.

Daly, C., R. P. Neilson, and D. L. Phillips. 1994. A digital topgraphic model for distributing precipitation over mountainous terrain. Journal of Applied Meteorology **33**:140–158.

Deeming, J. E., J. W. Lancaster, M. A. Fosberg, R. W. Furman, and M. J. Schroeder. 1972. National fire danger rating system. U.S. Forest Service Research Paper **RM-84**.

Kilgore, B. M. 1973. The ecological role of fire in Sierran conifer forests: its application to national park management. Quaternary Research **3**:496–513.

Kilgore, B. M. 1987. The role of fire in wilderness: a state-of-knowledge review. Pages 70–103 *in* R. C. Lucas, compiler. Proceedings of the National Wilderness Research Conference: issues, state-of-knowledge, future directions. U.S. Forest Service General Technical Report **INT-220**.

Kilgore, B. M., and D. Taylor. 1979. Fire history of a sequoia–mixed conifer forest. Ecology **60**:129–142.

McKelvey, K. S., C. N. Skinner, C. Chang, D. C. Erman, S. J. Husari, D. J. Parsons, J. W. van Wagtendonk, and C. P. Weatherspoon. 1996. An overview of fire in the Sierra Nevada. Pages 1033–1040 *in* Sierra Nevada Ecosystem Project: final report to Congress, volume II. Assessments and scientific basis for management options. University of California, Davis, Centers for Water and Wildland Resources, Davis, California, USA.

Miller, C., and D. L. Urban. 1999*a*. A model of surface fire, climate and forest pattern in Sierra Nevada, California. Ecological Modelling **114**:113–135.

Miller, C., and D. L. Urban. 1999*b*. Interactions between forest heterogeneity and surface fire regimes in the southern Sierra Nevada. Canadian Journal of Forest Research **29**: 202–212.

Miller, C. and D. L. Urban. *In press.* Connectivity of forest fuels and surface fire regimes. Landscape Ecology.

Moran, P. A. P. 1950. Notes on continuous stochastic phenomena. Biometrika **37**:17–23.

Mueller-Dombois, D., and H. Ellenberg. 1974. Aims and methods of vegetation ecology. John Wiley and Sons, New York, New York, USA.

Mutch, L. S. and D. J. Parsons. 1998. Mixed-conifer forest mortality and establishment before and after prescribed fire in Sequoia National Park, California. Forest Science **44**: 341–355.

Parsons, D. J. 1978. Fire and fuel accumulation in a giant sequoia forest. Journal of Forestry **76**:104–105.

Parsons, D. J., and S. H. DeBenedetti. 1979. Impact of fire suppression on a mixed-conifer forest. Forest Ecology and Management **2**:21–33.

Parsons, D. J., D. M. Graber, J. K. Agee, and J. W. van Wagtendonk. 1986. Natural fire management in national parks. Environmental Management **10**:21–24.

Pickett, S. T. A., and P. S. White. 1985. The ecology of natural disturbance and patch dynamics. Academic Press, Orlando, Florida, USA.

Rothermel, R. C. 1972. A mathematical model for predicting fire spread in wildland fuels. U.S. Forest Service Research Paper **INT-115**.

Ryan, K. C., and E. D. Reinhardt. 1988. Predicting postfire mortality of seven western conifers. Canadian Journal of Forest Research **18**:1291–1297.

Skinner, C. N., and C. Chang. 1996. Fire regimes, past and present. Pages 1041–1069 *in* Sierra Nevada Ecosystem Project: final report to Congress, volume II. Assessments and scientific basis for management options. University of California, Davis, Centers for Water and Wildland Resources, Davis, California, USA.

Smith, T. M., and D. L. Urban. 1988. Scale and resolution of forest structural pattern. Vegetatio **74**:143–150.

Stephens, S. L. 1995. Effects of prescribed and simulated fire and forest history of giant sequoia (Sequoiadendron giganteum [Lindley] Buccholz.)–mixed conifer ecosystems of the Sierra Nevada, California. Dissertation. University of California, Berkeley, California, USA.

Stephenson, N. L. 1996. Ecology and management of giant sequoia groves. Pages 1431–1467 *in* Sierra Nevada Ecosystem Project: final report to Congress, volume II. Assessments and scientific basis for management options. University of California, Davis, Centers for Water and Wildland Resources, Davis, California, USA.

Stephenson, N. L., D. J. Parsons, and T. W. Swetnam. 1991. Restoring natural fire to the sequoia–mixed conifer forest: should intense fire play a role? Pages 321–337 *in* Proceedings 17th annual Tall Timbers Fire Ecology Conference. Tall Timbers Research Station, Tallahassee, Florida, USA.

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

Urban, D. L., G. B. Bonan, T. M. Smith, and H. H. Shugart. 1991. Spatial applications of gap models. Forest Ecology and Management **42**:95–110.

Urban, D. L., C. Miller, P. N. Halpin, and N. L. Stephenson. *In press.* Forest gradient response in Sierran landscapes: the physical template. Landscape Ecology.

U.S. Congress. 1998. Forest Recovery and Protection Act of 1998, Report on HR 2515, House Report **105–440**, part 1, 105th Congress, second session. 12 March 1998.

Vale, T. R. 1998. The myth of the humanized landscape: an example from Yosemite National Park. Natural Areas Journal **18**:231–236.

Vankat, J. L. 1977. Fire and man in Sequoia National Park. Annals of the Association of American Geographers **67**: 17–27.

Vankat, J. L., and J. Major. 1978. Vegetation changes in Sequoia National Park, California. Journal of Biogeography **5**:377–402.

Van Wagner, C. E. 1973. Height of crown scorch in forest fires. Canadian Journal of Forest Research **3**:373–378.

van Wagtendonk, J. W. 1974. Refined burning prescriptions for Yosemite National Park. Occasional Paper 2. U.S. Department of the Interior, National Park Service, Washington, D.C., USA.

van Wagtendonk, J. W. 1985. Fire suppression effects on fuels and succession in short-fire interval wilderness ecosystems. Pages 119–126 *in* Proceedings of the symposium and workshop on wilderness fire. U.S. Forest Service General Technical Report **INT-182**.

van Wagtendonk, J. W. 1996. Use of a deterministic fire growth model to test fuel treatments. Pages 1155–1165 *in* Sierra Nevada Ecosystem Project: final report to Congress, volume II. Assessments and scientific basis for management options. University of California, Davis, Centers for Water and Wildland Resources, Davis, California, USA.

van Wagtendonk, J. W., J. M. Benedict, and W. M. Sydoriak. 1996. Physical properties of woody fuel particles of Sierra Nevada conifers. International Journal of Wildland Fire **6**: 117–123.

Wagener, W. W. 1961. Past fire incidence in Sierra Nevada forests. Journal of Forestry **59**:739–748.

Watt, A. S. 1947. Pattern and process in the plant community. Journal of Ecology **35**:1–22.