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Brandon M Collins* **and Scott L Stephens**

Past policies of excluding all wildfires from forests throughout the US are giving way to new strategies that incorporate naturally ignited fires into forest and fire management. In this paper, we evaluate the effects of long-standing natural fire programs (now referred to as wildland fire use or WFU) in two Sierra Nevada wilderness areas. We present reconstructions of historical fire occurrence using tree ring proxies, along with chronologies of tree recruitment, to infer the effects of WFU programs on forest structure. Historically, fires burned every 6 to 9 years, which moderated tree recruitment. Fire suppression policies established in the early 1900s successfully excluded fire and allowed for unprecedented tree recruitment. Despite the substantial changes in forest structure and composition, the frequency and extent of fires during the current WFU period (1972–present) approach historical levels. This information can provide some necessary insight in implementing WFU policy and developing management plans for similar forest types throughout the western US.

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Proposals that would allow naturally ignited wildfires to "run their course" have been seriously considered over the past several decades in the western United States. The premise of such arguments is that fire is a natural process and is necessary to maintain many forested ecosystems (Leopold *et al*. 1963; Houston 1971). For most forest types in this region, fire is a critical ecosystem process governing forest structure and composition at local and landscape scales. In the absence of frequent fire, drier mid- to low-elevation forests throughout the region have experienced increases in tree density, total biomass (fuel), and/or shifts in species composition (Parsons and Debenedetti 1979; Allen *et al*. 2002; Schoennagel *et al*. 2004). Additionally, fire exclusion has led to the homogenization of forested landscapes (Hessburg *et al*. 2005). These changes have not only altered forest community dynamics (ie light and water availability, nutrient cycling), but have also rendered extensive forested areas throughout the western US prone to exacerbated fire-induced effects (Agee 1998). It is necessary to note that the impacts of fire exclusion are less obvious in more mesic forest types, which have historically supported denser, closed-canopy structures (Schoennagel *et al*. 2004).

In recognizing the adverse impacts of fire exclusion on many forested ecosystems, and in compliance with the Wilderness Act (1964), which required wilderness managers to "preserve natural conditions", the US National Park Service adopted a policy in 1968 that included fire as a management tool to restore forested landscapes. This policy called for the use of both prescribed fire and natural fire (now referred to as wildland fire use, or WFU) to achieve restoration objectives. However, the uncertainties associated with natural wildfires coupled with the infrastructure and strong institutional dogma of fire suppression have limited the extent to which federal forest managers have implemented natural fire programs (Stephens and Ruth 2005).

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In recent years, exorbitant fire suppression costs, along with an increased understanding of the natural, or historical role of fire in western US forests, have forced land managers and policy makers to reconsider fire management. A guide to implementing WFU recently released by the US Federal Government states that, "fire, as a critical natural process, will be integrated into land and resource management plans and activities on a landscape scale and across administrative boundaries" (USDA/USDI 2006). This guide also explains that, in areas with approved fire management plans, WFU is to receive consideration and attention as a management action equal to wildfire suppression. If acted upon, these directives mark a fundamental shift toward incorporating natural wildfire into forest management. Here, we provide information that could be used in broader implementation of US WFU policy.

Where risk of escape is not excessively high and forecasted fire effects are desirable, allowing natural wildfires to burn not only restores ecosystem function, but reduces costs (eg suppression, restoration/fuel treatments) and improves firefighter safety (USDA/USDI 2006). Furthermore, WFU fires do not require the individual, intense planning process needed to implement prescribed burning and thinning projects (Ingalsbee 2001). As a result, WFU may be a more efficient strategy for managing large, remote areas of forested land. However, as rural and suburban communities expand into wildlands, and as smoke production from fires remains subject to the constraints of air quality regulations, WFU may continue to be limited geographically and temporally.

The few places in the western US that are relatively

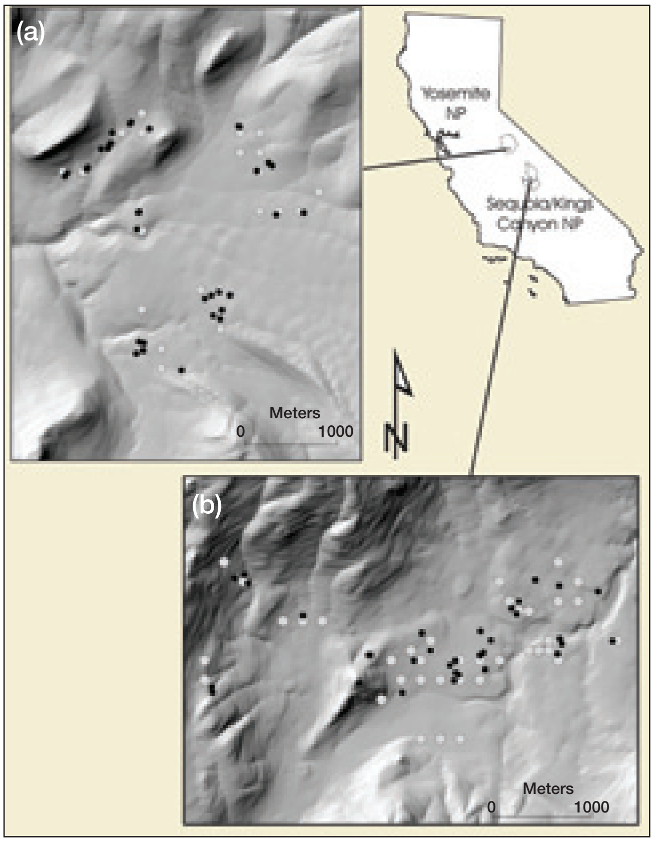

Figure 1. *Digital elevation models, stand/age structure plot (gray circles), and fire-scarred tree (black circles) locations in (a) Illilouette Creek basin and (b) Sugarloaf Creek Basin, CA.*

free from limitations on allowing natural fires are wilderness areas (Rollins *et al.* 2001). There are a handful of isolated wilderness areas in the region that have allowed lightning-ignited fires to burn for the past few decades. Illilouette Creek and Sugarloaf Creek basins in Yosemite National Park and Sequoia/Kings Canyon National Parks, respectively, are two such places (Figure 1). These basins provide a unique opportunity to evaluate the impacts, as well as the effectiveness, of over 30 years of WFU. Given the recent emphasis that federal land management agencies have placed on incorporating natural fire into their planning, an evaluation of long-standing WFU programs can provide necessary insight for the development of policy and management plans. To our knowledge, there has been very little ecological evaluation of WFU programs (but see Rollins *et al.* [2001] and Fule and Laughlin [2007]).

Here, we present reconstructions of historical fire occurrence using tree ring proxies, along with chronologies of tree recruitment, to infer the effects of WFU programs on forest structure. Our objectives were to compare the frequency and extent of WFU fires to that of historical (pre-fire-suppression) fires in Illilouette Creek and Sugarloaf Creek basins. In addition, we aimed to investigate the impact of the fire exclusion period on tree recruitment, relative to the historical time period and the WFU period in both basins. The term "tree recruitment" refers to trees that have established and persist to the present (sensu Brown and Wu 2005).

Methods

Illilouette Creek and Sugarloaf Creek basins are located in the central and southern Sierra Nevada range (CA), respectively (Figure 1). Each basin is over 15 000 ha, with elevations ranging from 1400 m to nearly 3000 m for the surrounding ridges. The climate is Medi-terranean, with cool, moist winters, and warm, generally dry summers. Between the two basins, average January minimum temperatures range from –2˚C to –5˚C, while average July maximum temperatures range from 24˚C to 32˚C. Precipitation varies with elevation and falls predominantly as snow, with annual averages near 100 cm in both areas. The forests in Illilouette Creek and Sugarloaf Creek basins are dominated by Jeffrey pine (*Pinus jeffreyi*), lodgepole pine (*Pinus contorta*), white fir (*Abies concolor*), and red fir (*Abies magnifica*), and are interspersed with meadows and shrublands.

We designated an approximately 500-ha study area within each basin (hereafter referred to as Illilouette and Sugarloaf) to investigate fire history and stand age structure. The locations of these study areas were chosen to optimally capture the range of area burned at different frequencies by WFU fires. We stratified the study areas by burn frequency (0–4 burns since 1972, when natural fire programs were initiated), then established a 200-m grid for stand age-structure sample locations. In Sugarloaf, we used a 100-m grid for the zero burn frequency stratum, because very little unburned area exists there. We intended to sample five plots in each burn frequency stratum, but, more importantly, we wanted to obtain ages from approximately 250 trees in each study area to study regeneration structure. Sixty-three 0.05-ha circular plots were sampled in total, with 24 of those in Illilouette and 39 in Sugarloaf. In each plot, we extracted increment cores from every live tree ≥ 10 cm diameter at breast height (dbh). Trees were cored, sometimes repeatedly, at approximately 20–30 cm above the ground level until we reached pith or no more than a field-estimated 10 years from pith.

We opportunistically collected cross-sectional slabs from 70 fire-scarred trees, snags, and downed logs. We cut and removed slabs from trees within approximately 70 m of plot center that exhibited visual evidence of multiple fire scars. We also collected slabs from any tree with multiple fire scars noticed as we walked among plots. Sampling only multiple-scarred trees results in an efficient method for detecting the maximum number of fire years with the least ecological damage and field/laboratory work (Brown and Wu 2005). Furthermore, recent research has shown that opportunistic or "targeted" sampling of fire-scarred trees to reconstruct historical fire occurrence yields results very similar to both random and systematic sampling (Van Horne and Fule 2006). Tree cores and fire-scarred slabs were sanded to a high polish, then crossdated against a master chronology using standard dendrochronological techniques to assign calendar years to pith dates and fire scars (methodology after Brown and Wu [2005]).

We limited the starting year of the analysis to 1700 because there was insufficient sample depth in the fire scar record prior to that period (Figure 2). We mapped the spatial extent of fires recorded within our study area by constructing a polygon around trees (three minimum) that recorded a particular fire (Bekker and Taylor 2001). While a complete census of all fire-scarred trees would be optimal for the most accurate reconstruction of fire extent, doing so would be impractical due to the effort required and the destructive nature of sampling fire scars, especially in wilderness areas. Based on our relatively complete coverage of each study area, we believe that we were able to sufficiently reconstruct fire extent (Figure 1). To investigate this assertion, we employed a species accumulation approach for analyzing fire years. We constructed accumulation curves by plotting the average number of fire years detected as a function of the number of fire-scarred trees sampled. The averages were calculated from 100 permutations, in which fire-scarred trees were added in random order up to the total number of trees for each study area (n = 33 and 37 for Sugarloaf and Illilouette, respectively). Michaelis-Menton saturating functions were fit to each curve to estimate the asymptote, or theoretical maximum number of fire years, for each study area (Battles *et al*. 2001). The accumulation curves (not shown) indicate that, in both study areas, the total number of fire years detected is within 20% of the theoretical maximum for Illilouette and 26% for Sugarloaf. We believe that this discrepancy is reasonable, given that nearly half of all fire years detected in both areas appear to be localized "spot" fires that scarred two or fewer trees. Therefore, we have probably detected all widespread fires.

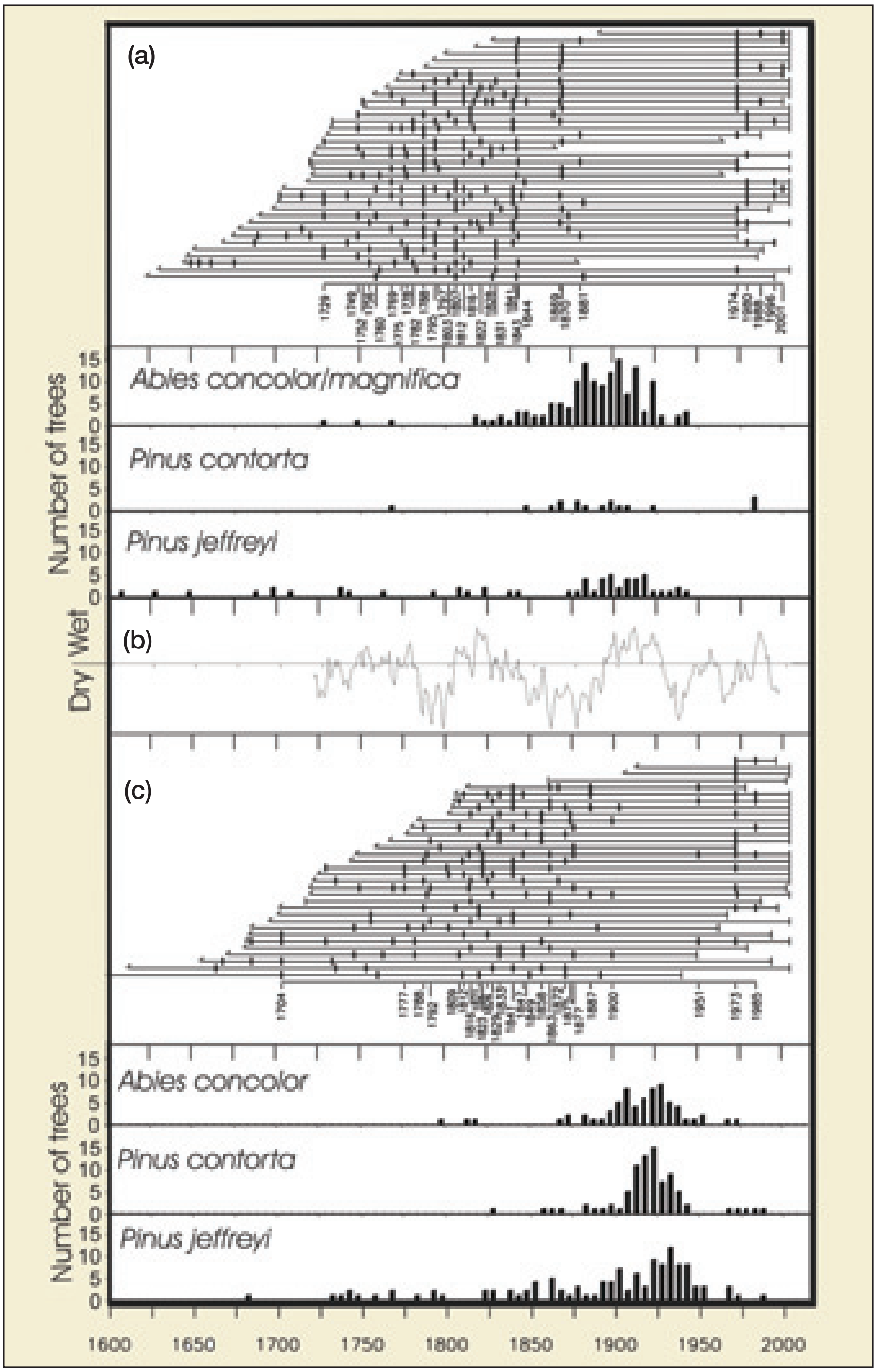

*Figure 2. Fire year and tree recruitment chronologies for (a) Illilouette Creek and (c) Sugarloaf Creek basins. Horizontal lines represent the timespan of individual fire-scarred trees, with dark vertical tick marks representing fire scars. Listed below are fire years in which ≥3 trees were scarred. Solid vertical bars below each fire chronology are tree recruitment dates by 5-year periods. (b) Twenty-year moving average of reconstructed Palmer Drought Severity Index for grid point 14 (*Cook et al. *1999).*

Results

We were able to establish pith dates for 489 trees and to successfully crossdate 420 fire scars from 70 trees between Illilouette and Sugarloaf (Figure 2). We could not determine pith dates on 22 of 239 trees in Illilouette and 28 of 300 trees in Sugarloaf. In Illilouette, white fir was the most abundant

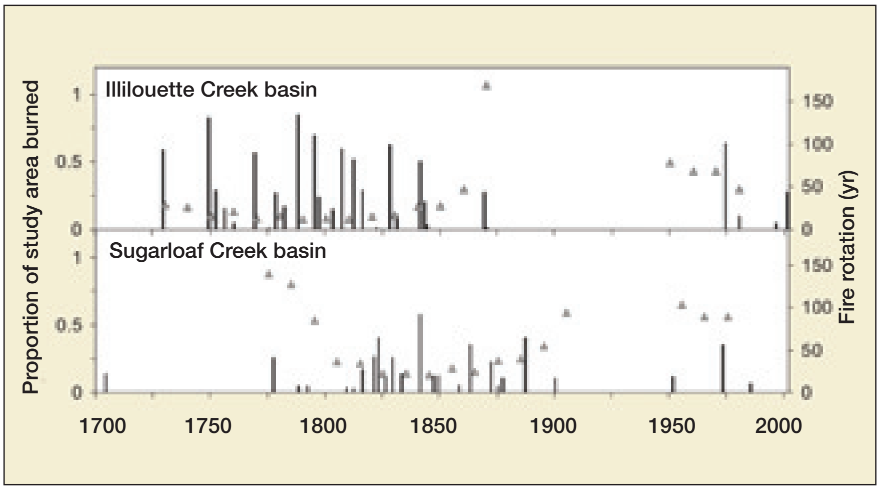

Figure 3. Reconstructed fire extent (vertical bars) and fire rotation (gray triangles), within each study area, for years in which ≥3 trees were scarred. Fire rotation is defined as the length of time necessary to burn a cumulative area equivalent to the size of the study area, and is calculated at 10-year intervals for overlapping 50-year periods.

tree species (60%), followed by Jeffrey pine (26%), lodgepole pine (8%), and red fir (6%). The mix of tree species was more even in Sugarloaf, with Jeffrey pine being most abundant (44%), followed by lodgepole pine (31%) and white fir (25%). In both areas, the oldest trees were Jeffrey pine, which exceeded 300 years in age, while white and red fir and lodgepole pine seldom exceeded 200 years. White and red fir recruitment and, to a lesser extent, lodgepole pine and Jeffrey pine recruitment, noticeably peaked between 1875 and 1920 in Illilouette. In Sugarloaf, a similar peak in recruitment is evident for all three species; however, this peak occurred later, between 1895 and 1940 (Figure 2).

The earliest recorded fires were 1650 for Illilouette and 1665 for Sugarloaf. Fires occurred fairly frequently throughout the reconstructed period of record (Figure 2). Using a minimum criterion of at least three trees recording a given fire year, the mean fire return interval (MFI) between 1700 and 1900 was 6.3 years for Illilouette and 9.3 years for Sugarloaf. Using the same minimum criterion and time period, the fire rotation (defined as the length of time necessary to burn a cumulative area equivalent to the size of the study area) was 24.7 years for Illilouette and 49.2 years for Sugarloaf. This difference is due to the generally smaller fires in Sugarloaf (Figure 3). Fires clearly stopped occurring after 1881 in Illilouette and after 1904 in Sugarloaf, with the last fires of any substantial spatial extent occurring in 1869 and 1900 in Illilouette and Sugarloaf, respectively (Figures 2 and 3). With the exception of the 1951 escaped wildfire in Sugarloaf, fires did not occur again until the start of WFU policies. For the WFU period (1972–present), we detected five fires, with an MFI of 6.8 years in Illilouette, while only two fires were identified in Sugarloaf, with a 12-year interval between the fires. Fire rotations during this period were 32.9 and 79.7 years for Illilouette and Sugarloaf, respectively.

Discussion

As in many of the dry forests throughout the western US, fire has historically played a major role in shaping and maintaining the forests in Illilouette and Sugarloaf (Figure 2). The frequency at which fires burned, along with the persistence of many of the older trees (mostly Jeffrey pine) suggests that fires in the 200 years prior to fire exclusion typically burned under moderate to low intensities. Based on the data presented here, there is little evidence to suggest WFU fires burned differently. The frequency and extent of fires during the current WFU period approaches that of historical levels, especially in Illilouette (Figure 3). In fact, analysis of recent large WFU fires in both Illilouette Creek and Sugarloaf Creek basins demonstrates that spatial patterns of fire-induced mortality are similar to such patterns for historical fires, as we understand them (Collins *et al.* 2007).

Similarities between historical fires and recent WFU fires are surprising, given the obvious changes in tree recruitment that coincided with the fire exclusion period (Figure 2). In both Illilouette and Sugarloaf, unprecedented peaks in tree recruitment began shortly after the last extensive fire in each area (1869 in Illilouette and 1887 in Sugarloaf). The cessation of fires, along with a shift toward wetter climatic conditions around 1900, most likely explains the observed pulses in tree recruitment (Brown and Wu 2005; Figure 2). Historically, frequent fires moderated tree recruitment by killing small trees that had not grown to the size at which trees resist fire (ie tree crowns were not high enough off the ground or bark was not thick enough to insulate tissues from thermal damage; North *et al.* 2005; Stephens and Fry 2005). The long fire-free interval resulting from fire exclusion allowed for increased establishment and growth beyond this vulnerable stage for all three dominant tree species in Sugarloaf, and for mostly white fir in Illilouette. While we do not know the extent to which trees were killed by WFU fires, especially early, larger WFU fires, we do know that many survived these fires and persist to the present.

Given that so many of the trees which became established during fire exclusion survived multiple WFU fires, it seems probable that they will continue to persist for some time (Miller and Urban 2000), barring the occurrence of much more severe fires in the future. While it is possible that competitive interactions among these trees have influenced, and continue to influence, forest structure, we submit that fire is the dominant process-driving structure in these forests. As such, restoring historical forest structure by fire alone may not be feasible for the near future. However, what may be more important than restoring structure is restoring the process of fire (Stephenson 1999). By allowing fire to resume its natural role in limiting density and reducing surface fuels, competition for growing space would be reduced, along with potential severity in subsequent fires (Fule and Laughlin 2007). As a result, we contend that the forests in Illilouette and Sugarloaf are becoming more resistant to ecosystem perturbations (eg insects, disease, drought). This resistance could be important in allowing these forests to cope with projected changes in climate.

Ultimately, active restoration to historical forest structure across landscapes may be undesirable, considering the effort required to manipulate tree density and fuels. Furthermore, the climate of the historical period is a poor predictor of future climates (Millar *et al.* 2007). However, there is clearly a need to incorporate fire into the management of drier, historically more open forest types throughout the western US. The results from this study show two clear examples of such a forest type, in which the process of fire has successfully been returned over each respective landscape. Although it is not ubiquitously applicable, WFU could potentially be a cost-effective and ecologically sound tool for "treating" large areas of forested land. Decisions to continue fire suppression are politically safe in the short term, but ecologically detrimental over the long term. Each time the decision to suppress is made, the risk of a fire escaping and causing damage (social and economic) is essentially deferred to the future. Allowing more natural fires to burn under certain conditions will probably mitigate these risks. If the public is encouraged to recognize this and to become more tolerant of the direct, near-term consequences (ie smoke production, limited access) managers will be able to more effectively use fire as a tool for restoring forests over the long term.

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References

Agee JK. 1998. The landscape ecology of western forest fire regimes. *Northwest Sci* **72**: 24–34.

Allen CD, Savage M, Falk DA, *et al.* 2002. Ecological restoration of southwestern ponderosa pine ecosystems: a broad perspective. *Ecol Appl* **12**: 1418–33.

Battles JJ, Shlisky AJ, Barrett RH, *et al.* 2001. The effects of forest management on plant species diversity in a Sierran conifer forest. *Forest Ecol Manag* **146**: 211–22.

Bekker MF and Taylor AH. 2001. Gradient analysis of fire regimes in montane forests of the southern Cascade range, Thousand Lakes Wilderness, California, USA. *Plant Ecol* **155**: 15–28.

Brown PM and Wu R. 2005. Climate and disturbance forcing of episodic tree recruitment in a southwestern ponderosa pine landscape. *Ecology* **86**: 3030–38.

Collins BM, Kelly NM, van Wagtendonk JW, and Stephens SL. 2007. Spatial patterns of large natural fires in Sierra Nevada wilderness area. *Landscape Ecol* **22**: 545–57.

Cook ER, Meko DM, Stahle DW, and Cleaveland MK. 1999. Drought reconstructions for the continental United States. *J Climatol* **12**: 1145–62.

Fule PZ and Laughlin DC. 2007. Wildland fire effects on forest structure over an altitudinal gradient, Grand Canyon National Park, USA. *J Appl Ecol* **44**: 136–46.

Hessburg PF, Agee JK, and Franklin JF. 2005. Dry forests and wildland fires of the inland northwest USA: contrasting the landscape ecology of the pre-settlement and modem eras. *Forest Ecol Manag* **211**: 117–39.

Houston DB. 1971. Ecosystems of national parks. *Science* **172**: 648–50.

Ingalsbee T. 2001. Wildland fire use in roadless areas: restoring ecosystems and rewilding landscapes. *Fire Manag Today* **61**: 29–32.

Leopold AS, Cain SA, Cottam CM, *et al.* 1963. Wildlife management in the national parks. *Audubon* **65**: 166–71, 184–85.

Millar CI, Stephenson NL, and Stephens SL. 2007. Climate change and forests of the future: managing in the face of uncertainty. *Ecol Appl*. In press.

Miller C and Urban DL. 2000. Modeling the effects of fire management alternatives on Sierra Nevada mixed-conifer forests. *Ecol Appl* **10**: 85–94.

North M, Hurteau M, Fiegener R, and Barbour M. 2005. Influence of fire and El Niño on tree recruitment varies by species in Sierran mixed conifer forest. *Forest Sci* **51**: 187–97.

Parsons DJ and Debenedetti SH. 1979. Impact of fire suppression on a mixed-conifer forest. *Forest Ecol Manag* **2**: 21–33.

Rollins MG, Swetnam TW, and Morgan P. 2001. Evaluating a century of fire patterns in two Rocky Mountain wilderness areas using digital fire atlases. *Can J Forest Res* **31**: 2107–23.

Schoennagel T, Veblen TT, and Romme WH. 2004. The interaction of fire, fuels, and climate across Rocky Mountain forests. *BioScience* **54**: 661–76.

Stephens SL and Fry DL. 2005. Spatial distribution of regeneration patches in an old-growth *Pinus jeffreyi*–mixed conifer forest in northwestern Mexico. *J Veg Sci* **16**: 693–702.

Stephens SL and Ruth LW. 2005. Federal forest-fire policy in the United States. *Ecol Appl* **15**: 532–42.

Stephenson NL. 1999. Reference conditions for giant sequoia forest restoration: structure, process, and precision. *Ecol Appl* **9**: 1253–65.

USDA/USDI (US Department of Agriculture/US Department of the Interior). 2006. Wildland fire use implementation procedures reference guide. Washington, DC: USDA/USDI.

Van Horne ML and Fule PZ. 2006. Comparing methods of reconstructing fire history using fire scars in a southwestern United States ponderosa pine forest. *Can J Forest Res* **36**: 855–67.

