History and Management of Crown-Fire Ecosystems: a Summary and Response

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Introduction

Some ecosystems, such as yellow pine forests, have had a long history of frequent surface fires, but because of fire suppression policy, fires have been largely excluded from them during the last century (Covington 2000). Unnatural fuel accumulation in these forests has increased the potential for large, catastrophic crown fires, and reintroduction of prescribed fire is one remedy for this critical fire hazard. But fire ecologists and fire managers need to be cautious in transferring this model to all western ecosystems (Anderson et al. 1999; Gutsell et al. 2001). Although large, catastrophic crown fires are apparently unnatural in yellow pine forests (but cf. Shinneman & Baker 1997), this is not so in other western forests and shrublands, and widespread prescription burning is not warranted everywhere.

Johnson et al. (2001 [this issue]), illustrate how this yellow-pine model has been inappropriately applied to the boreal forests of Canada, where crown fires are an inevitable consequence of fuel structure and burning is not age-dependent. Thus, creating a landscape-age mosaic with prescription burning will not reduce the incidence of crown fires in these forests (Johnson & Miyanishi 1995).

The yellow-pine model has also been misapplied to the chaparral shrublands of southern California, which has led to the erroneous conclusion that large crown fires in these ecosystems are a modern artifact of fire-suppression policy (Minnich 1983, 1989, 1995, 1998). However, large crown fires predate fire-suppression activities in California (Keeley & Fotheringham 2001 [this issue]) and other shrublands (Pyne 1991; Gill 2000). Also, it has been argued that the solution to preventing large, catastrophic fires is the use of widespread prescription burning to create a landscape mosaic of different-aged patches of vegetation (Minnich 1989, 1995, 1998). This fuel-age/mosaic model does not fit California shrublands, however, because fire-suppression policy has not resulted in fire exclusion, and there has been as much or more area burned by wildfires in recent decades than before active fire suppression (Moritz 1997, 1999; Conard & Weise 1998; Keeley et al. 1999). Failure to effect fire exclusion results from the fact that fire management is challenged with an ever-increasing rate of fire incidence which parallels the exponential rate of human population growth (Keeley 2001) in an environment with the worst fire weather in the country (Schroeder et al. 1964). An important consequence is that there has not been an unnatural accumulation of fuels. Recent studies show that fire hazard is either independent of age (Moritz 1999) or only weakly dependent up to 20 years of age (Schoenberg et al. 2001); large, catastrophic fires will readily burn through young stands and do not require old vegetation (Fig. 1). In short, there are sufficient data to refute the contention that chaparral "fire occurrence is constrained in space and time by the rate of fuel accumulation and by previous fire history" (Minnich 2001 [this issue]). Any constraints are weak at best.

The theoretical basis behind the fuel-age/mosaic model lies in earlier modeling work done on California shrublands (Rothermel & Philpot 1973; Philpot 1974). Based on fire-spread models, it was concluded that as chaparral stands increase in age, there is a resultant increase in fuels, rate of fire spread, and fire size. Following suggestions by Countryman (1974), these models were interpreted to support a fire-management policy that relied heavily on prescription burning to produce a landscape comprising a mosaic of age classes. The thinking was that as fires burn across a landscape and encounter patches of younger age classes, they either die out because of insufficient fuels or their spread

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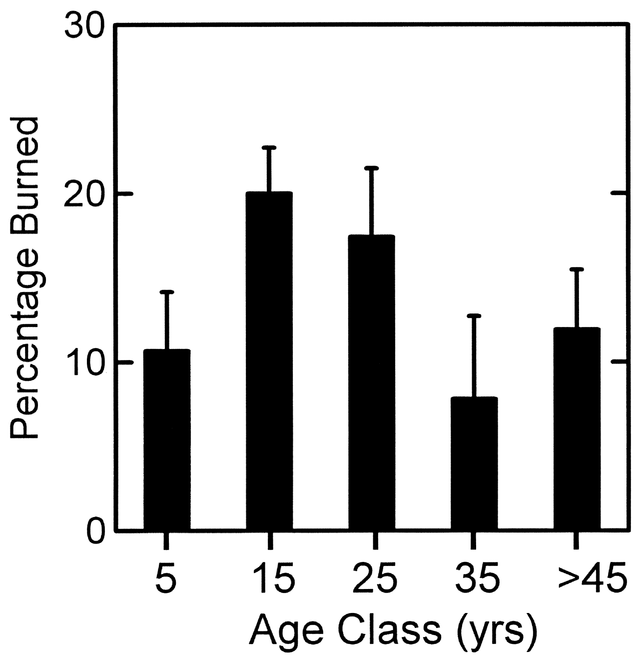

Figure 1. Vegetation age classes burned in the eight largest fires during the 30-year period of 1967–1996 in the shrub-dominated Santa Monica Mountains (Ventura and Los Angeles counties, California) (data from Keeley et al. 1999).

rates decline, making them more amenable to fire-suppression activities.

There are several problems with applying this fuel-age/mosaic model in California shrublands. One is the assumption that fire suppression has excluded fire and resulted in a chaparral landscape glutted with an unnatural accumulation of old age classes. This assumption is false across much of southern California, where chaparral stands younger than 35 years are the norm (Keeley et al. 1999). Another limitation is that the model assumes spread rates based on fire behavior documented from prescribed burns, which show that stands <20 years of age will not carry fire (Green 1981). But prescribed burns are conducted under low to moderate fire-weather conditions. In contrast, fire-behavior studies of wildfires under severe weather conditions demonstrate that chaparral crown fires will burn through any age class (Fig. 1), and even young stands in the path of such fires are hazardous sites for attacking fires (Countryman 1974; Dunn & Piirto 1987). Also, the validity of these early models is called into question because they were based on the incorrect assumption that fuels increase in a predictable cumulative fashion (Payson & Cohn 1990; Conard & Regelbrugge 1994; Regelbrugge 2000). Most important, modeling studies that consider landscape patterns of fire spread conclude that stand age alone cannot constrain fire size (Zedler & Seiger 2000).

Thus, in California shrublands, landscape-scale age mosaics may affect the size of fires ignited under moderate weather conditions, but they are not effective barriers to fire spread under severe weather conditions. Considering the issue of loss of human lives and property, it is the fires that burn under severe conditions that are most destructive, and landscape-scale age mosaics do not pose an effective barrier to such catastrophic fires. These conclusions are shared by others confronted with a similar shrubland fire hazard in South Africa (Brown et al. 1991) and Australia (Buckley 1992; Bradstock et al. 1998*b*).

Johnson et al. (2001) argue that in all closed-canopy ecosystems such as boreal forest and chaparral, large fires are weather-driven phenomena. The same appears to apply to Douglas-fir/western hemlock forests of the U.S. Pacific Northwest (Agee 1997), red cedar forests of the northern Rocky Mountains (Habeck 1985), and sclerophyllous shrub/woodlands of Australia (Pyne 1991; Gill 2001). The widely publicized Cerro Grande fire, which burned in yellow-pine forests around Los Alamos last year, was driven by severe fire weather, suggesting that weather is an important determinant in the yellow-pine fire regime as well. Agee (1997) argues, however, that weather is critical in these forests only because fire suppression has all but eliminated natural fires and caused an unnatural accumulation of fuels. Minnich (2001) likewise argues that severe weather is relevant in chaparral only because of fire suppression. Agee's argument is well grounded in the fact that, in the yellow-pine forest, fire suppression equals fire exclusion, but this is not the case in chaparral.

An ancillary issue is the historical role of fire, something that reflects on the extent to which prescription burning is needed for maintaining these fire-dependent communities (i.e., burning for resource benefit). Johnson et al. (2001) contend that prescription burning has no role in the boreal forest because the current fire-return interval is lower than the average lifetime of the dominant trees. Likewise, fire is not in short supply in the sclerophyllous shrub/woodlands of Australia; rather, high fire frequencies, often caused by prescription burning, pose a clear extinction risk to many species (Whelan & Muston 1991; Bradstock et al. 1998*a*). In coastal California shrublands, historical analyses fail to justify prescription burning for resource benefit because a fire-suppression policy has not grossly altered fire regimes (Keeley & Fotheringham 2001 [this issue]). Not only is ecosystem health not at risk from lack of fire, but the massive number of anthropogenic ignitions poses a clear threat to species that are near their limits of resilience or have already been extirpated from sites (Zedler et al. 1983; Haidinger & Keeley 1993; Zedler 1995). As pointed out by Gill (2001 [this issue]), this creates a dilemma when one is forced to choose between reducing fire hazard through frequent prescription burning and biodiversity conservation.

Fire Regime of Baja California

A peripheral issue addressed in the lead article of this forum is what can be learned by contrasting shrubland fire regimes north and south of the U.S.–Mexican border. Dodge (1975) pioneered this approach, and Minnich (1983, 1989, 1998) has used this comparison to argue that the pattern of small fires south of the U.S. border is a model of what fire regimes were like in southern California prior to fire suppression. Minnich argues that if managers could recreate this pattern it would prevent large fires. But patterns of burning in Baja California are of limited value in reconstructing natural fire regimes north of the border, not only because of biogeographical differences but because they are strongly influenced by different land-use patterns. Baja California has many times more anthropogenic fires than does southern California. Limited fire prevention south of the U.S. border, coupled with intensive land-based resource exploitation (Henderson 1964; Ojeda et al. 1991; Bullock 1999), has produced a landscape that bears little resemblance to the natural condition. Human impact is evident in the extensive distribution of exotic annual grassland, which invariably is the vegetation that links urban areas with native shrublands in a seemingly increasing zone of disturbance (see the vegetation map given by Minnich and Franco-Vizcaíno [1998]). Such conversion of shrubland to exotic grassland not only reflects high disturbance but contributes to further increases in fire incidence (D'Antonio 2000).

In support of the Baja model, Minnich (2001) presents a new figure contrasting the age structure of chaparral stands in San Diego County and adjacent Baja California, as they appeared 30 years ago. This is a colorized version of data presented earlier (Minnich 1989, 1995, 1998), and there are problems with his methodology and interpretation. This figure focuses on a much smaller subset of the landscape than the original Landsat study (Minnich 1983). It purports to include only chaparral, but there are large blocks of grassland, oak savanna, and coniferous forests included north of the border (c.f. vegetation map of County of San Diego [1977]). Excluding these areas makes the coverage more comparable to that shown south of the border, and doing this breaks up many of the large age-class blocks into smaller fragments, more similar to the pattern south of the border. More important than this, however, is the way in which this map has been interpreted (Minnich 1989, 1995, 1998, 2001; Minnich & Chou 1997). This type of age-class map is not appropriate for making inferences about fire size, because any contiguous area mapped as the same 5-year age class could comprise a composite of adjacent fires that burned in any one of the 5 years. Also, the methodology lacks any means of evaluating the accuracy of these determinations. South of the border, a 52-year fire history was reconstructed from three aerial photographs, and no controls were presented to show that aerial photographs taken 16–18 years apart were sufficient to accurately map 5-year age classes. Because fire history north of the border was based on written records, the appropriate control for the Baja data would have been to determine the extent to which aerial photographs north of the border taken 18 years apart could accurately map 5-year age classes. This sort of groundtruthing is necessary to place statistical bounds around the conclusions drawn from remote images. For example, Minor (1989) reported on the ability of Landsat images (same technology utilized by Minnich 1983) to detect same-year fires and reported that their conclusions were off by 40% or more. Skepticism here is justified by the fact that there are reports of large fires in northern Baja California (e.g., Plummer 1911; Henderson 1964; Haiman 1973; Amaya 1991) that have either been missed or downplayed.

Equally important are questions about the data set used north of the border, which was heavily biased against small fires; those of <16 ha (40 acres) were not included. Minnich and Chou (1997) attempted to correct this bias by excluding small fires from their reconstruction of fire history for Baja California. As a consequence, however, we do not have a true picture of fire size either north or south of the border, because an extraordinary proportion of the fires have been excluded. California Division of Forestry (CDF) records for San Diego County show that over 95% of all the recorded wildland fires (CDF 1970–1979) were <16 ha; thus, Fig. 1 of Minnich (2001) reflects fewer than 5% of all fires north of the border and an unknown proportion south of the border.

One of the strongest arguments against the fuel-age/mosaic model as an explanation for the Baja burning patterns is the calculated fire-rotation interval, or what Minnich (2001 [this issue]) terms the "turnover of fire patches." This is reported to be 60–80 years, and to make these numbers compatible with his fuel-age model, he assumes that chaparral stands of <60 years of age fail to carry fire (Minnich 1989, 1995, 1998; Minnich & Chou 1997). Empirically, we know that fires, even under moderate weather conditions, readily carry through chaparral far younger than this (Green 1981; Biswell 1989). It appears that northern Baja California is a mosaic of many small burns that are dispersed within a sea of very old chaparral stands (see Fig. 1 of Minnich 2001 [this issue]). The reason fires do not spread cannot be simply a function of fuel age and is likely a function of a number of factors, including biogeography, topography, climate, and land-use patterns (Keeley & Fotheringham 2001 [this issue]).

Other regional comparisons of fire regimes illustrate the importance of biogeographical differences, in particular the role of climate. For example, Heyerdahl et al. (2001) show that U.S. ecosystems spanning a similar latitudinal range to that considered in Minnich's papers, but with no difference in fire-suppression policy, exhibit very different fire regimes between the northern and southern extremes. Forests at the southern end of the range exhibit higher fire frequencies and earlier burning seasons, similar to the patterns reported for Baja California (Minnich 1983, 1989, 1995). Weather is a strong determinant of fire regimes in crown-fire ecosystems (Johnson et al. 2001 [this issue]), and in southern California chaparral one of the more important weather patterns is the autumn winds known as Santa Anas, which decline in importance south of the border (Keeley & Fotheringham 2001 [this issue]). Minnich (2001) disputes this and claims that such winds are well represented south of the border, but are called "El Nortes." Although Santa Ana and El Norte winds have some similarities, they are not synonymous phenomena: the latter are desert winds restricted to the Gulf of California (Badan-Dangon et al. 1991; Godsey 2001). As for Santa Ana winds, we would not argue that they have been denied an entrance visa to cross the border, but we do contend that they diminish southward, being largely absent from the southern half of the Baja region considered by Minnich (1983).

Minnich (1989, 1998, 2001) states, based on little data, that immediately south of the U.S. border, fires driven by Santa Ana winds are nonexistent and he claims that this reflects the effectiveness of the fuel-age/mosaic model. Our skepticism of this explanation is based on the observation that, in southern California, fires driven by Santa Ana winds will readily burn through young age classes (Fig. 1). In response to this observation Minnich (2001) now argues that Santa Ana winds fail to drive fires in northern Baja because such winds occur too late in the season and thus are "constrained by patches produced by summer burns." For this to be true, one would have to believe that every summer a sufficient number of small patches throughout northern Baja California are burned to act as fuel breaks for the subsequent autumn fires driven by Santa Ana winds which otherwise might have occurred that same year. This seems improbable and inconsistent with a fire-rotation interval of 60–80 years. We doubt Minnich's assertion that fires driven by Santa Ana winds are absent immediately south of the border, but if true, a more likely explanation is that autumn burning is inhibited by the Mexican Monsoon (Douglas et al. 1993) (e.g., a 24-year record for Aseradero at 1580 m in the Sierra Juarez shows an average July and August precipitation of 33 mm and 40 mm, respectively [S. Reyes, unpublished data]), whereas comparable elevations north of the border have at least an order of magnitude less precipitation during the summer months (National Oceanic and Atmospheric Administration 1999).

Source of ignition is highly variable between regions, and in this respect crown-fire ecosystems may be quite different from one another. In the boreal forests, lightning coincident with a high-pressure system is the primary source of ignition (Nash & Johnson 1996). In contrast, on the shrub-dominated landscapes of southern California, annual lightning-strike density (Minnich et al. 1993) is an order of magnitude less than in the boreal forest, and humans account for 95% of all fires (Keeley 1982). Consequently, we have some trouble accepting the claim that northern Baja California is saturated with natural sources of fire ignition and that people have had no effect on the incidence of fire (Minnich 1995, 1998, 2001). This case for lightning saturation has been overstated. The calculations presented by Minnich (2001) contain a 10-fold conversion error; based on Minnich et al. (1993), one should expect a 1-km^2 patch to be struck by lightning only once every 10 years. Minnich et al. (1993) estimate that roughly 1 out of every 50 strikes results in fire, so, on average, a 1-km^2 patch of shrubland in Baja California is struck by lighting once every 10 years but is ignited by lightning only once every 500 years. Minnich and Chou (1997) contend that this incidence of lightning fire is sufficient to account for the frequency of fires in their data, but they forget that their data do not include fires of <16 ha and thus are a weak indicator of fire incidence on the Baja landscape. North of the border the flux rate of lightning strikes is comparable to that of Baja (Minnich et al. 1993; M. Wells, personal communication), and written records show that for every lightning-ignited fire, humans ignite 20 more (Keeley 1982). Minnich and Chou (1997) report that there are 7.7 times more fires south of the border, and this increase appears to be due to human subsidy. Thus, north of the border humans have had a substantial effect on fire incidence, and south of the border this effect appears to be many times larger.

Historical Fire Records

Minnich (2001) argues that twentieth-century fire data are inappropriate for detecting fire-suppression effects. He contends that the miniscule fire-suppression forces present at the end of the nineteenth century were sufficient to alter the natural fire regime, but this seems disingenuous because he maintains repeatedly that contemporary fire-suppression efforts in Baja California are completely ineffective. In support of his belief that fire suppression has been effectively excluding fires for well over a century, he cites the decline in fires during the late nineteenth century reported for giant sequoia forests (Swetnam 1993). Indeed, fire-scarred conifers throughout the southwestern United States and Mexico show a diminished fire frequency in the final decades of the nineteenth century, but this is generally attributed to either loss of Native American burning or to removal of fine fuels by livestock grazing (e.g., Savage & Swetnam 1990; Fulé and Covington 1999). Using twentieth-century historical records to examine fire suppression effects, and the assumption that suppression activities were more intense in the second half of the century (Moritz 1997; Conard & Weise 1998; Keeley et al. 1999), are both justified by the fact that the average annual area burned in the United States declined an order of magnitude after 1950 (Dombeck 2001). Regardless of how one draws the historical trajectory of fire-suppression effectiveness, the important conclusion from fire-history records is that the yellow-pine-ecosystem equation of fire suppression equals fire exclusion is not applicable to California shrubland landscapes.

Conclusions

Too many unanswered questions remain about the factors determining the fire regime of Baja California to make it a useful model for the historic pattern in California or as a model for guiding U.S. fire-management policy. We believe that fire-management policy is best guided by contemporary analysis of fire behavior, in chaparral and other crown-fire ecosystems. In both boreal forests and chaparral, the yellow-pine model is inappropriate, and large, catastrophic crown fires are less dependent on unnatural accumulation of fuels and more dependent on ignitions coincident with severe weather. In these ecosystems, the widespread application of prescription burning to create age mosaics is not cost-effective management (Bradstock et al. 1998*b*; Conard & Weise 1998; Johnson et al. 1998). In southern California, and perhaps in other regions as well, we recommend management that focuses on strategic placement of prescribed burns and more serious consideration of other options (e.g., U.S. Congress 1958; Omi 1979; Zedler 1995), with particular focus on constraining the rapidly expanding urban-wildland interface.

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