

Fire Frequency in Southern California Shrublands: Biological Effects and Management Options

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Abstract. Much is known about fire effects on shrublands but there are strong disagreements about the historical role of fire and the proper use of fire in management. Studies of species responses to fire at the population level are one way to gain a better understanding of the effect of fire frequency. Three risks to shrubland species from extremes of fire return interval are identified: senescence risk, immaturity risk, and recruitment risk. The first is posed by long intervals between fires, and the last two by short intervals. It is argued that for San Diego County, at least, only immaturity risk appears serious at this time. Consequently, from a strictly biological point of view, the idea that there is an urgent need to reduce the time between fires cannot be supported. A brief review of historical data on fire occurrence also suggests that expectations on the part of managers for a landscape in which catastrophic fires cannot occur may be unrealistic. It is acknowledged that management for purely biological objectives is not possible, and present landuse policies make it nearly impossible to allow fire to play a more natural role. The only solution may be to concentrate efforts along the urban-wildland boundary while working to reduce unnecessary intrusions into wildlands that make management difficult.

Keywords: Chaparral; Coastal sage scrub; fire; fire history; fire management; plant life history; shrublands

Introduction

There is universal agreement that fire is an integral part of the ecology of the shrublands of southern California, but there is no clear consensus on either the history or the desired future of shrubland fire regimes. This applies with special force to the critically important question of the length of time between fires and its inverse, fire frequency.

There is disagreement on this critical issue because it is difficult to get clear, unequivocal answers to

questions about the effects of alternative fire regimes. Large-scale experimentation is logistically challenging, costly, and most discouragingly, requires at least decades to produce clear answers. Historical data can provide valuable insight, but even if we knew the history of fire in great detail, past fire regimes cannot, in their entirety, be imposed on our present landscape which has been irretrievably altered by development and the introduction of exotic species.

It is the premise of this paper that much can be learned by focusing attention on the population ecology of the shrubland species, and by making use of wildfires as well as experiments to observe how populations respond to different intervals between fires. I will try to show that it is possible to get a sense for at least the limits of fire frequency beyond which large changes may be expected in the vegetation. The questions addressed are these: 1) How resilient are plant species to variations in the return time of fire? 2) What is the response of plant species of different life history to extremes in fire return time? 3) What are the implications of individual species response to fire management?

Sufficient data has accumulated that a substantial review could have been written. For reasons of convenience, however, these questions will be addressed by examples drawn from research and observations in the shrublands of San Diego County, California.

Fire Regime

By the broadest definition, fire regime includes all the characteristics of the fires, such as their intensity and rate of spread and their size and occurrence in time and space. Though the size and shape of burns can be important to the population as a whole, to the individual plant and the local population it is intensity of the fire (heat release per unit time per unit fire front length) and the length of time that has passed since the last fire that

are of greatest importance. The first determines the direct physical effect of the fire on the plant, and the second the characteristics and location of the plant tissues affected. Intensity effects will be considered in another paper in this volume. Here, the importance of the time since the last fire, or fire return time, will be the primary variable considered.

Fire frequency, as the number of fires per unit time, is often used as the descriptor of the return time aspect of fire regime. But it is probably not the best one. In physics, frequency of the electromagnetic spectrum is the number of cycles per unit time, and period is its inverse. For fire, the inverse of frequency is the time between fires (also "fire return time" or "fire interval"). But confusion arises because frequency is also the common term for number or proportion of observations in a particular class or at a particular value. This usage sometimes produces implicit or explicit statements about fire frequency that do not have the meaning given above (e.g., Minnich 1989, p. 42; Johnson 1992, p. 80). Though the term "fire frequency" is too well-entrenched to be supplanted, it aids clarity to speak in terms of time between fires whenever possible.

It is only in theory that the time between fires can be constant, and therefore to describe a fire regime adequately, we must know both the mean time between fires and something about the variability in time between fires. Fire return is therefore best thought of as a random variable with a probability distribution (Johnson 1992). The greatest degree of elegance is obtained when the fire regime can be defined by mathematical models, allowing means, variances, and other useful statistics such as the probability of fire interval exceeding a specified value to be calculated (Johnson 1992).

In reality, data do not fit simple theoretical models precisely, nor are the parameters constant over time. As a further complication, actual runs of intervals between fires are likely to show autocorrelation because climatic cycles appear to be important in fire regimes (Johnson 1992, Swetnam 1993). Many believe that changes in human population densities and landuse practices also change the probability distribution of recurrence times by changing the number, place, and seasonal distribution of ignitions. Considering the great changes that have occurred in climate (Cole 1983, Singh 1988), human population, and landuse practices over the last 10,000 years, it is doubtful if the probability distribution of fire recurrence has been stable for more than very short periods. This observation anticipates a conclusion of this paper that the species of the fire-prone shrublands must possess considerable resilience to variation in the time of occurrence of fire.

Chaparral Fire Succession

The mechanisms by which plant species of the southern California shrublands recover from fire are well known and well documented (e.g., Cooper 1922, Horton and Kraebel 1955, Vogl 1970, Vogl and Schorr 1972, Keeley and Zedler 1978, Westman et al. 1981, Biswell 1989). As Hanes (1971) noted, the capacity to recover from fire is so pronounced, that dispersal into burns is, or was in the natural landscape, of scant importance, and succession after fire involves primarily the germination of dormant seeds and the stimulation of sprouting in surviving plants. Because this succession does not involve a sequence of invasions of plants from outside the area of the disturbance, Hanes (1971) argued that it deserved the special term "autosuccession" which underscores the fact that recovery is by propagules and individuals present on the site before the fire.

Chaparral post-fire succession can therefore be simplified to a process in which fire releases individuals (ramets and genets) from dormant seed and bud "banks" (Fig. 1).

The budbank is the primary means of recovery by shrubs and perennial herbs. The dormant seedbank is a critical means of survival for many species. In fact, if each seed is considered an individual, then for many species most of the population consists of dormant seeds. For shorter-lived shrubs (*Lotus scoparius*, *Helianthemum scoparium*) and a large number of herbaceous annuals, the only individuals at the time of the fire will be those in the seedbank. A few species, most notably *Adenostoma fasciculatum*, have both a large budbank and a large seedbank, but most reestablish primarily by one means or the other.

To facilitate discussion, it is useful to distinguish "seedlife," the lifespan of a seed from maturation to germination or death, and "plantlife" as the life span of an individual genet (individual established from seed) from germination to death. Because of the extremely high mortality characteristic of the seedling stage of most plants, a further refinement might be to define "mature plant life" as the average life span of individuals that reach reproductive maturity. The relative length of plantlife and seedlife varies significantly among plants of the chaparral. For the non-seeding sprouter *Quercus berberidifolia*, for example, with acorns susceptible both to drought and herbivory, the mature plantlife/seedlife ratio is certainly >100 , whereas the post-fire early successional species *Lotus scoparius* has a ratio that is very probably <1 as it is short-lived but with a very persistent seedbank.

Consideration of these basic life history features suggests where shrubland fire succession may be sus-

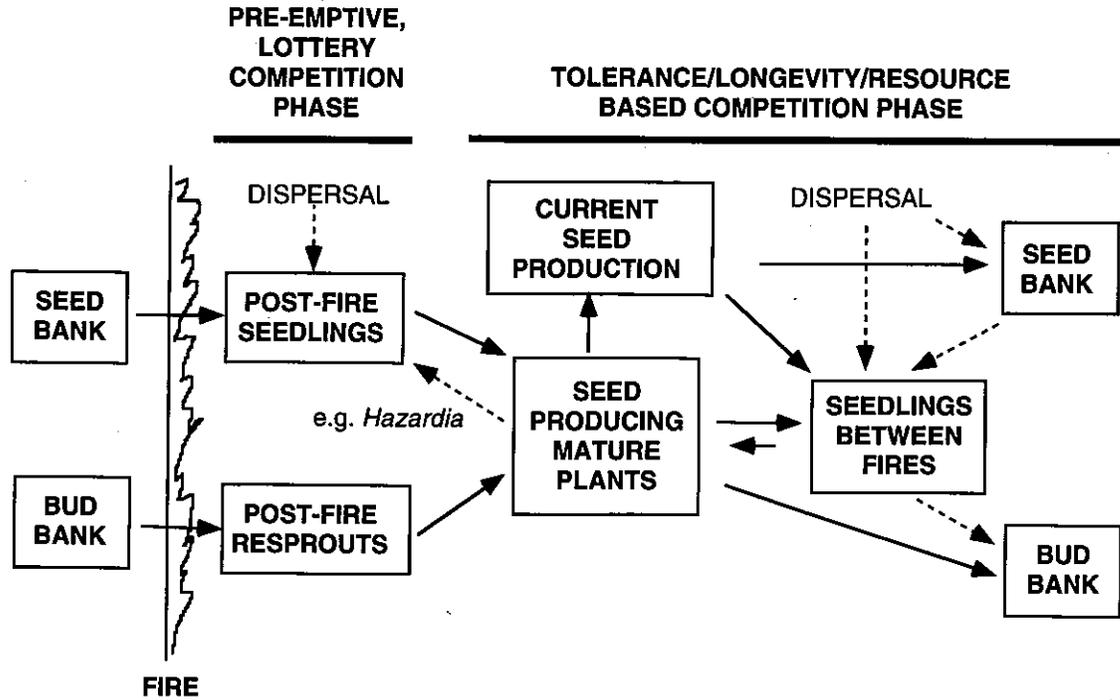


Figure 1. Simplified diagram of successional processes in shrublands subject to fire. The two phases of competition favor different life history strategies.

ceptible to disruption. One possibility, which will not be considered in this paper, is that extreme intensity will kill the dormant buds and seeds, thus decreasing or eliminating the regenerative capacity of species. Because of the dearth of quantitative intensity measures, however, it is difficult to generalize about its importance. Intensity, because it is partly dependent on the amount of fuel accumulated, will also to some extent be a function of the interval between fire. Some believe that fire suppression has caused fires to be more intense and therefore more destructive. But the rate of biomass accumulation tends to level off with age, and therefore it is likely that age is a poor predictor of intensity past some relatively young age (Black 1987). Intensity effects will not be considered here, but they cannot be dismissed as unimportant.

The time between fires, however, is more easily determined, and can potentially have profound effects. At one extreme, long intervals between fire may be destructive for species that germinate in abundance only after fire. If these intervals are longer than (plantlife)+(seedlife) for those species that sprout after fire or longer than (seedlife)+(age at last reproduction) for non-sprouting seeding species that are killed by fire, then the populations could suffer local extinction. Because these potentially dire consequences are a result of population senescence before a rejuvenating fire, this potential problem may be called "senescence risk".

At the other extreme, fire may recur before a species that depends on its seedbank has had an opportunity to accumulate a sufficient seed reserve. This will inevitably be the case if the interval between fires is less than age at first reproduction. Sprouting species may also suffer if all dormant buds have been exhausted and no new ones produced by the next fire or if all energy reserves have been expended and none remain to support resprouting. These hazards collectively may be called "immaturity risk."

Another possibility, only recently brought forward for serious consideration, is that short intervals between fires may limit recruitment opportunities for non-seeding sprouters (Keeley 1992b). This could occur if shrublands failed to develop litter layers, lacked certain plant-determined microhabitats or faunal assemblages, or failed to develop soil properties that favor seedling establishment between fires. The difference between this risk and immaturity and senescence risk is that the longevity and vigor of the sprouters would make it difficult to detect change after a single fire. Failure of recruitment would be apparent only after there had been significant mortality of mature plants, a process that would take at least decades, and for some species, centuries. To emphasize the importance of the germination microhabitat, the term "recruitment risk" will be used.

Sensitivity of a Non-sprouting Serotinous Conifer, Tecate Cypress, to Variations in Fire Recurrence Interval

Seed storage on the plant in fire-resistant structures is one of the most important ways in which plants deal with fire in mediterranean-climate regions of Australia and South Africa (Lamont et al. 1991, van Wilgen and Forsyth 1992). This mode of coping with fire is limited in California to a few species of conifers in the genera *Cupressus* and *Pinus* which form woodlands closely associated with shrublands but are usually classified as distinct from them (Vogl et al. 1977). One of these species, Tecate cypress (*Cupressus forbesii*) occurs in scattered groves from Orange County into Baja California (Griffin and Critchfield 1972, Minnich 1987). Because of its short stature, dense canopy, and association with shrublands, it may be considered a conifer-dominated phase of the shrublands with a fire regime that is probably more controlled by the time of burning of the adjacent shrublands than by any of its intrinsic properties.

The life history of cypress with respect to fire is simple (Fig. 2). Seeds are released from cones on the tree after fire and there is generally a high percentage of germination. Seedling establishment is all but unknown at other times except where there has been human disturbance. Because of the severe sites on which the cypress grows, the canopy in cypress stands is always close to the ground and more or less continuous with the canopies of co-occurring shrubs. Intense crown fires invariably are the result. Resistance of the trees to fire is poor, and even those on the margins of fires whose crowns are unburned often succumb simply from the heat shock. They are incapable of sprouting from the base.

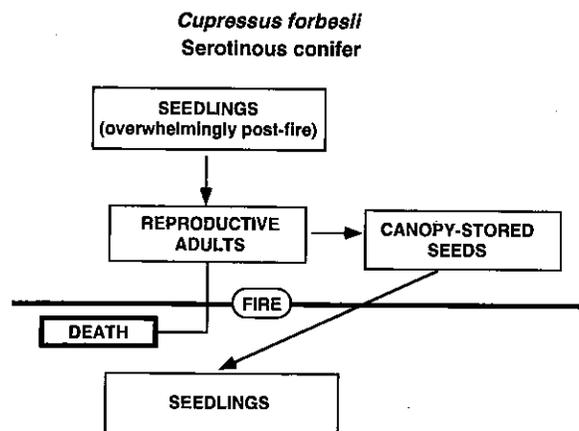


Figure 2. Life history of Tecate cypress. All individuals touched by fire are killed and regeneration is strictly from seed.

With this life history, survival through a fire is completely dependent on the accumulation of viable seeds in the serotinal cones. Such a life history is clearly susceptible to immaturity risk, but could also be exposed to significant senescence risk if individuals died before the next fire.

Data on seed crops observed in stands of different ages (Fig. 3) show that cypress cone accumulation is slow to begin, but that it continues over a long period of time. Since the number of seeds is proportional to the number of cones, it can be seen that stands unburned for over 90 years have substantial seed reserves. But seed reserves do not reach high numbers until stands are 20-40 years old.

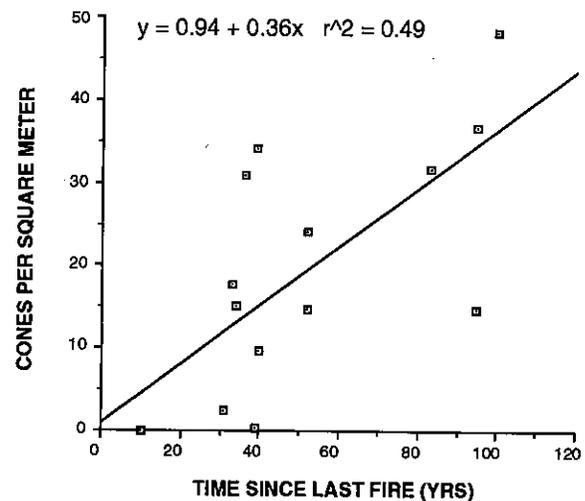


Figure 3. Accumulation of cones on cypress trees with increasing age for stands on Otay and Tecate Mountains, San Diego County, California. There is no evidence for a decline in cone crop with age. Zedler, unpublished data.

The ultimate test of the sensitivity of the species is the consequences to stands of being burned at different times. As has been noted in Zedler (1981) and Dunn (1987), repeated burning of stands on Tecate Mountain lead to the local extinction of the species in some areas, and substantial reduction in population size and density in others. The data in Fig. 4 show that the stand in this set that was less than 20 years old failed to reestablish, whereas older stands fared much better and in general reoccupied the site at densities that were sufficient to insure dominance of the site.

These data suggest a species that is capable of enduring through long fire-free periods. While it would seem probable that at some age the population could become senescent, the age at which this occurs is certainly greater than 100 years. Based on the longevities of related species, survival in good condition to 200 or more years is likely. Thus the senescence risk, while

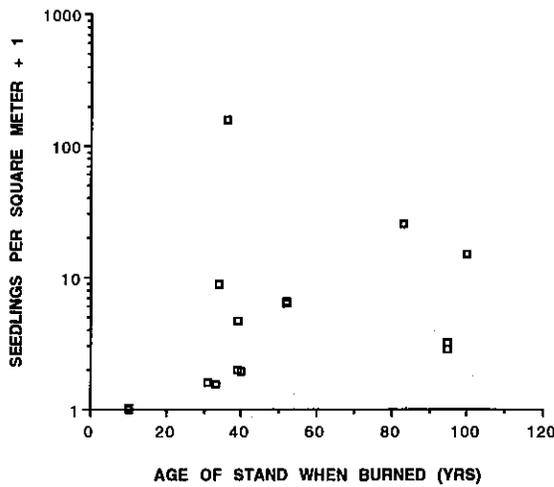


Figure 4. Observed densities of seedlings in cypress stands burned at different ages. Data from Otay and Tecate Mountains, San Diego County, California. Zedler, unpublished.

probably present, is not a problem now and will not be for another century. In contrast, the immaturity risk is significant, as evidenced by the population reductions that have been observed.

Senescence Risk in Non-sprouting Seeding *Ceanothus* spp. : Plant and Seed Survival in *C. greggii*.

Unlike Tecate cypress, which is an edaphic endemic of only local importance, *Ceanothus greggii* is both geographically widespread and often locally dominant or co-dominant. The life history of *Ceanothus greggii* is very similar to that of Tecate cypress (Fig. 5),

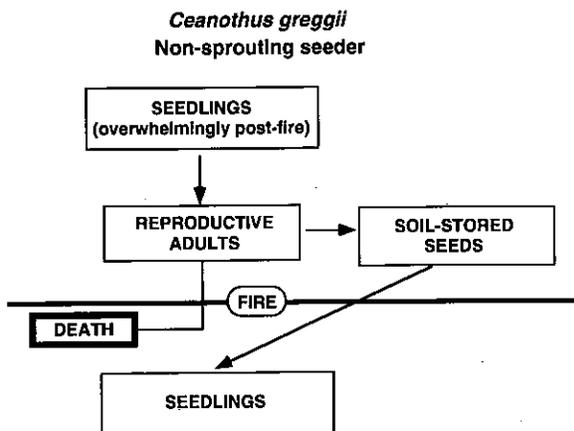


Figure 5. Life history of *Ceanothus greggii*. It is essentially identical to that of Tecate cypress except that its dormant seeds are in the soil.

but it differs in two important features. First, though *C. greggii* is killed by fire and reestablishes from seed, it does so with seeds that are present in the soil rather than on the plant. Second, the life expectancy of *C. greggii*, though not known with precision, is much shorter than that of Tecate cypress, and it is common to find dead or dying individuals in chaparral of all ages. In very old stands, a majority of individuals, including those in the largest size classes, will be dead (Fig. 6, and Zammit and Zedler 1993).

The risks to *C. greggii* would be the same as those faced by Tecate cypress. The senescence risk, however, would seem to be much more acute, since the average longevity of individuals is far less. Studies across an

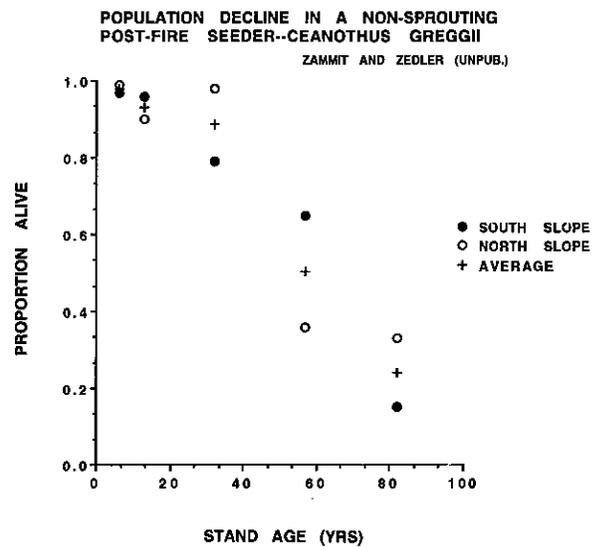


Figure 6. Decline in abundance of *Ceanothus greggii* with increasing age of stands as indicated by the proportion of the population that consists of live individuals. Though populations at the greatest ages have only a few live individuals, these continue to flower and produce seeds. From Zammit and Zedler (1993), and more important, the seedbank is at levels not significantly different from young stands (Fig. 7).

age gradient have established, however, that even though populations have been depleted in old stands (Fig. 6) seed production per plant continues at levels comparable to young stands,

Thus the senescence risk, though undoubtedly present, according to this data set would not be significant up to fire return intervals of 100 years. This prediction of good recovery capacity has been confirmed when one of the old stands in which the data were obtained was burned when control lines failed to contain a management burn in the summer of 1992. Seedlings of *C. greggii* were recorded at a density sufficient to replace the mature population assuming usual survivorship (DeSimone, Cario, and Zedler, un-

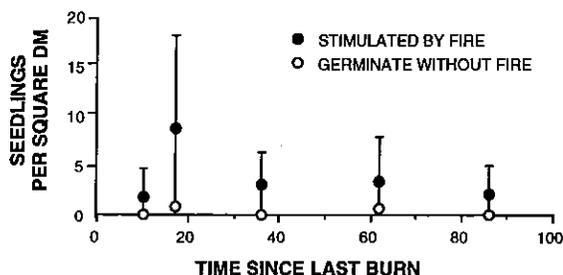


Figure 7. Germinable seedbank of *C. greggii* in stands of different ages. Variability is high, but there is no evidence for a sharp decline with age. Zammit and Zedler (1993).

published; Mills and Kummerow 1989). This example supports the idea that senescence risk for *C. greggii* may not become serious until well after 100 years.

Immaturity Risk in *Ceanothus*: Elimination of *C. tomentosus* from Large Areas by Fires at Short Intervals.

As in Tecate cypress, a delay before a substantial seed reserve accumulates would make non-sprouting *Ceanothus* species liable to disastrous decline if stands were burned at a young age. An actual occurrence of this theoretical possibility was reported by Zedler et al. (1983). The situation arose when ryegrass was seeded into chaparral and coastal sage scrub on Otay Mountain after a wildfire in 1979 in 36 year old vegetation. The take of the ryegrass was unusually good, and an arson fire set in the summer of 1980 reburned a substantial portion of the 1979 burn area. The shrublands within this burn therefore were subject to two fires only a year apart, a circumstance not impossible in natural conditions, but clearly in this specific instance attributable to human intervention.

C. tomentosus, though in a different sub-genus from *C. greggii*, has the same general life history. The consequences to the populations of *C. tomentosus* within the area that was burned twice were devastating (Fig. 8). The large number of seedlings established from the seedbank that had accumulated in the 36 years between fires proved to be very sensitive to fire, and, like adults, to be incapable of resprouting. As a result, *C. tomentosus* was eliminated from large areas. This is a demonstration that short intervals between fires can pose a serious threat to non-sprouting seeders. For this species, at least in human-influenced landscapes, immaturity risk is clearly very great.

Senescence Risk in *Adenostoma fasciculatum*

A. fasciculatum is the most abundant species of the California shrublands. Like *C. greggii*, it is both widespread and often locally very abundant. Unlike *C. greggii*, *A. fasciculatum* commonly forms almost pure stands over hundreds or even thousands of hectares. Its life history combines variable but usually strong-to-moderate sprouting after fire with the copious production of seeds that will germinate after fire but will in lower numbers also germinate at other times (Fig. 9). *A. fasciculatum* has been said to be a short-lived, and subject to massive die-off in old stands (Hanes 1981). Our research in San Diego County has not confirmed this. It is true that *A. fasciculatum* individuals in stands that have been unburned for 50 or more years generally have numbers of dead stems of all sizes, including the largest. It is also true that many individuals of *A. fasciculatum* die during the course of stand develop-

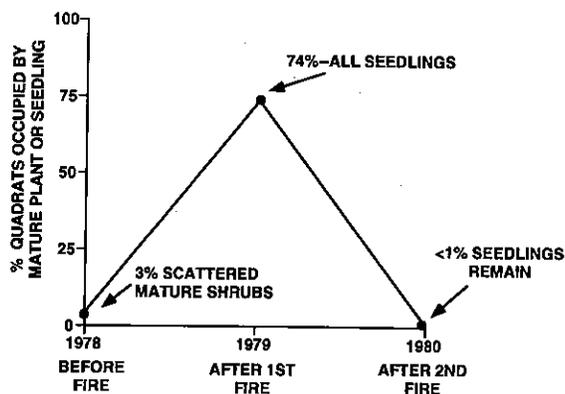


Figure 8. The catastrophic decline in *Ceanothus tomentosus* resulting from two fires a year apart on Otay Mountain, San Diego County, California. From Zedler et al. 1983.

ment after fire. Those established from seed seem especially susceptible to competition. Overall, however, the stands remain viable with high seed production and an ability to produce sprouts from the base to replace stems killed in fire.

Further, as with *C. greggii*, seed production remains high, and seedbanks are large. For our data set, in fact, the maximum levels of seedbank were recorded in one of the oldest stands (Fig. 10). It is probably not the case that seedbanks necessarily always increase with age, but there is nothing in these data to support the idea of a significant senescence risk to *A. fasciculatum*.

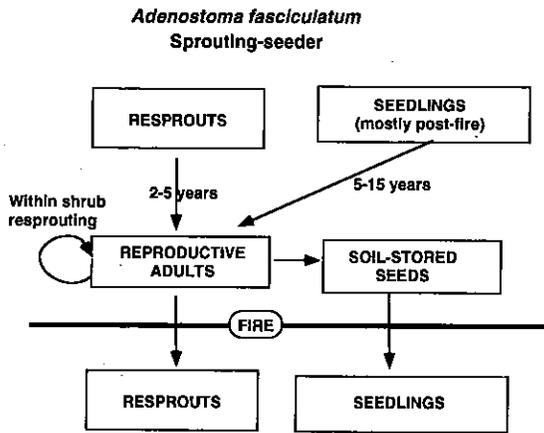


Figure 9. Life cycle of *Adenostoma fasciculatum*.

Immaturity Risk in *Adenostoma fasciculatum*

In a species that both sprouts and seeds after fire, immaturity risk can be significant only if both seedling establishment and resprouting is significantly impaired. Surprisingly, under some circumstances, *A. fasciculatum* populations can be decimated or virtually eliminated over large areas by fires occurring at short intervals. In the 1979-80 sequence of fires on Otay Mountain described above, the consequences of this short interval fire were almost as devastating to the *A. fasciculatum* populations as they were to the *C. tomentosus* (Fig. 11) to populations. As is usual, *A. fasciculatum* both sprouted and established many seedlings after the 1979 fire. The grass fire, though of relatively light intensity, neverthe-

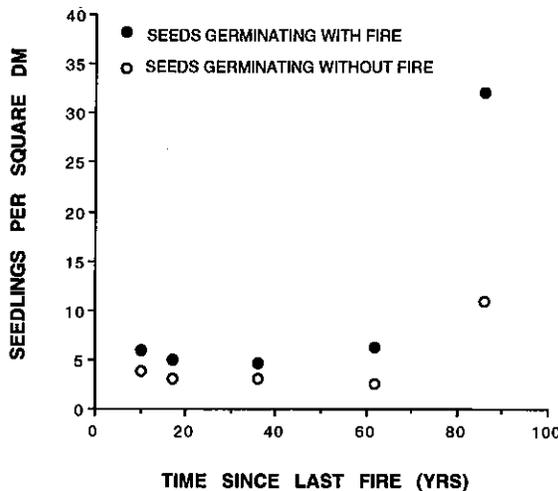


Figure 10. Seedbanks of *A. fasciculatum* as estimated by germination experiments. Data from Zammit and Zedler 1993.

less killed nearly all seedlings and also killed a large proportion of the resprouting individuals. Thus, despite two means of reproduction, *A. fasciculatum* had a substantial immaturity risk with this short interval between fires. How long after a fire *A. fasciculatum* populations remain at risk from reburns is unknown.

Non-seeding Sprouters

For the large number of species of non-seeding sprouters, senescence risk would be large only if longevities were less than fire return intervals. For the majority of these species, longevity is unknown, but is clearly substantial, certainly exceeding 100-200 years for most species. Included here are chaparral species

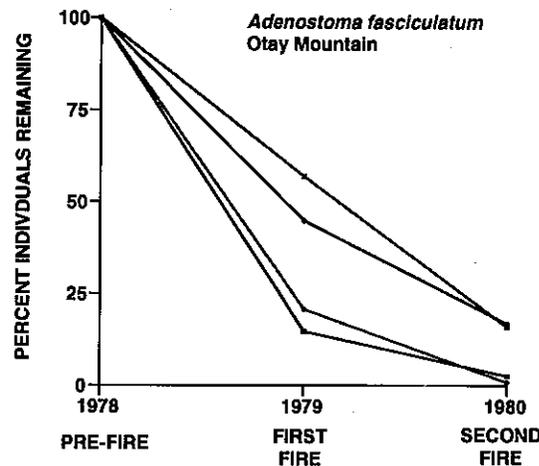


Figure 11. Reduction in *A. fasciculatum* after two fires separated by only a year. The four lines are replicate sites. Data from Zedler et al. (1983).

such as *Quercus berberidifolia*, *Heteromeles arbutifolia*, *Rhus ovata*, *Cecrocarpus betuloides*, *Rhamnus crocea*, *R. ilicifolia*, and *Arctostaphylos glandulosa*; and coastal sage scrub and coastal chaparral species such as *Rhus integrifolia*, *Cneoridium dumosum*, *Comarostaphylis diversifolia*, and *Xylococcus bicolor*. This life history pattern is also known from other mediterranean-climate areas (Trabaud 1987, Lloret and López-Soria 1993).

Though comments about the senescence, decadence, over-maturity, or trashy quality of old shrubland stands with these species are often made informally, I know of no convincing evidence that members of this group of species have any significant senescence risk, with or without fire protection. On the contrary, available evidence starting with Horton and Kraebel (1955), supported by our unpublished data, and confirmed recently by Keeley (1992a) indicates that most, if not

all, of these species increase with time in unburned chaparral by the establishment of seedlings beneath the canopies of other species. Most also are capable of increase in stature and canopy breadth with increasing time between fires. Thus, I rate the senescence risk for most of these species as low-to-non-existent.

The example of *Adenostoma fasciculatum*, however, suggests that immaturity risk is a possibility if they are susceptible to reburn in the early stages of resprouting. Available data suggests that this risk, too, is very low. In the Otay 1979-80 fire discussed above, one of the stands studied was on a north-facing slope and had a large component of non-seeding sprouters (including *Xylococcus bicolor*, *Cercocarpus minutiflorus*, and *Quercus berberidifolia*). The grass fire, though it killed many of the sprouts, had little effect on mortality of individual shrubs, which, unlike many *A. fasciculatum* on nearby sites, were capable of resprouting a second time (Zedler et al. 1983). Presumably, recovery was retarded to some extent, but it is doubtful that the fact that this stand burned twice in two years could be determined today by analysis of shrub density or size.

Discussion

What are the limits of shrubland resilience when subjected to varying fire return intervals?

The data presented above suggest that the immaturity risk is far greater than the senescence risk. No data were presented on "recruitment risk", but because it can be significant only for long-lived sprouters, if it exists, its effects will not become serious for many decades or centuries. Therefore, from a biological point of view, no strong argument can be made for decreasing the interval between fires. The often-heard assertion that fire suppression threatens the health and survival of the chaparral by increasing the interval between fires has little or no support in studies of the response of individual species. In contrast, short intervals between fires are known to sometimes have very pronounced effects.

Another conclusion that can be drawn from examples presented here is that most species in the shrublands of southern California do possess considerable resilience to variations in time of fire. For example, the seedbank of *Ceanothus greggii* observed across the chronosequence seems sufficient to permit recovery of populations with ages ranging from 20 to 100 years. There seems little reason to assume that shrubland species can tolerate fire recurrence over only a very narrow range — the life histories of species

seem designed to accommodate the vicissitudes of fire regimes. But although a shift in fire regime may not often lead to local extinction of species, it could well cause substantial shifts in relative abundance leading, over long periods of time, to competitive displacements. Studies are needed to explore these possibilities more fully.

These conclusions do not directly contradict the idea widely held by managers that suppression has artificially lengthened the interval between fires. They do, however, erode one of the pillars supporting it. Fire managers often state that controlled fires are needed to "rejuvenate" shrublands. Often one hears that artificial burns are "returning fire to its natural role in the landscape." The implication is that populations are at peril unless burned "often," whereas the bulk of the data suggests otherwise.

Are past fire regimes a model for the present?

Another prop of the belief common among managers that current fire regimes have deleteriously long intervals between fire is the conviction that the historical record shows that fires of the past were a more frequent, smaller, and altogether friendlier phenomenon. Thus, it is argued, controlled fire is necessary to reduce intervals between fire, and return us to those Arcadian days. One accepting this view could argue that a failure to show disastrously bad consequences to long intervals between fires is not a sufficient argument against using controlled fires to decrease the intervals between fires. In fact, the apparent resilience of shrubland species suggests that fire interval could be reduced substantially, provided that the immaturity or recruitment risks are not significantly increased. The historical record, however, with the exception of areas apparently managed by native Americans, does not support the view that past fires were smaller and recurred more frequently.

The most comprehensive study of wildfire patterns of the recent past for southern California is that of Minnich and his colleagues (Minnich 1989). Minnich has mapped fire size and date of occurrence in California and adjacent Baja California and shows that north of the U.S. border there are more large fires and fewer small fires. South of the border, where fire suppression is not a high priority, fires are smaller and more numerous. It is argued that the Mexican pattern is closer to the pre-European or natural pattern, whereas the U. S. pattern of fires reveals the bad effects of fire suppression, which sets up the vegetation for large catastrophic fires. A surprising finding very relevant to the arguments of this paper is that despite the great differences in spatial pattern, the turn-around time of

fire (the total area studied divided by the average area burned each year) is about 70 years in both regions, suggesting that the fire return intervals in the two regions may not be so different and that the intervals on average are quite long. To many managers, 70 year old shrublands are "senescent." This raises a dilemma for managers who subscribe to the need for rejuvenation of decadent chaparral: if they accept Minnich's conclusion that a fine-grained mosaic is natural, they should also accept a 70 year rotation time, at least for chaparral. The data presented here suggest that that would be acceptable from a biological point of view, but it is unlikely to meet fuel reduction goals.

If we make the assumption that the turn-around time estimated from Minnich's data is the average return time of fire, it would seem that for the species discussed in this paper both immaturity risk and senescence risk are minimal on both sides of the border. But averages are not the whole story. The two instances of immaturity risk discussed here, the shrubs on Otay Mountain and the cypress on Tecate Mountain, both describe situations within the study area that Minnich considers. Thus, at least locally, some populations suffer from short intervals between fires. It could be, however, that these are anomalies. That both of these cases involved arson fires, however, makes one uneasy that human ignitions, expected to increase with increasing population density, may pose an increasing danger to natural vegetation.

But other studies call into question the conclusion that large catastrophic fires were rare in the past, even as the studies support the conclusion that fire-return times probably exceeded 30 years. An independent study of fire history by Byrne and his colleagues provides a basis for evaluating the idea that catastrophic fires were not a part of the pre-European landscape and, most relevant to this paper, also gives another estimate of probable times between fires (Byrne et al. 1977, Byrne et al. unpublished report). This work is based on the occurrence of charcoal in layered sediments in the Santa Barbara Channel from which the annual rates of charcoal deposition can be determined and followed back for hundreds of years. The interpretation of the data is not entirely straightforward. There are complexities involving the source contributing carbon, the transport mechanisms, and the residence time of the charcoal during transport to the site of deposition. Nonetheless, several important conclusions emerge. First, contrary to the expectation that fires are larger now than in the past, a comparison of the 1931-1970 (i.e., post-suppression) record with the pre-European period of 735 A. D. to 1505 A. D. shows that charcoal was deposited much more uniformly in the recent past than in pre-European times. The pre-Euro-

pean record shows large peaks separated by long intervals of much lower deposition, a pattern that could only have been produced by the occasional occurrence of very large fires. Thus, catastrophic fire is not a 20th century invention (see also Zivnuska 1977).

It is more difficult to deduce fire-free intervals from the charcoal records. The main problem is that the charcoal deposited in the channel comes from a large area and from vegetation of varying biomass. Thus, the years between major peaks do not necessarily reflect return times, because the areas burned may not have overlapped, and peaks of equal magnitude do not necessarily come from burns of the same size. Thus, when the authors state that "major fires occurred in the southern (coastal) area of the Los Padres National Forest on average once every 65 years" and "further inland fires occurred more frequently, on average once every 30-35 years" (Byrne et al. unpublished), this does not mean that the fire return times at points on the ground necessarily were 65 and 30 years. Because charcoal input between peaks is continuous, it is possible that they were less than these figures suggest. It is more likely that they were substantially more, because the peaks in charcoal were probably produced by fires that overlapped little or not at all. Though we cannot be certain of the return time from these data, it is clearly inappropriate to conclude that a 30-35 year fire rotation in controlled burns will simulate the natural pattern.

Another question concerns the impact of Native Americans on the vegetation. Historical, cultural, and some independent data (Byrne, unpublished) point to the frequent use of fire by the Native Americans. It seems quite probable that certain coastal and mountain areas were actively managed by fire (Anderson 1993). The grassy areas reported by early explorers along the coast (e.g. Crespi in Bolton 1927), if as free of woody plants as some accounts implied, probably could not have been maintained by lightning alone, but rather were maintained by frequent fires of human origin. Too much has been lost for us to know the details of this presumed fire management, but we cannot assume that it was inconsequential.

Choosing a fire regime

In discussing management goals it is important to separate biologically-based considerations from those related to human health and safety and the protection of property. A biologically-based management scheme does what is appropriate to maintain the natural systems. In contrast, a management scheme that aims at minimizing economic and social damage will not be concerned about the degradation of natural systems if

this has no economic impact. It would be nice if the two sets of goals led to the same conclusions about management policies, but it would be a mistake to assume that they will.

From the biological perspective, appropriate fire management maintains biodiversity — it keeps all of the species, plants and animals, in the system while minimizing the abundance of exotics. I suggest that we concentrate suppression efforts on areas recently burned, discontinue controlled burning except for the probably rare instances where the need for fire could be clearly demonstrated in very old shrub vegetation, and continue programs aimed at discouraging arson. Wildfires, whatever their origin, should be allowed to burn unless shown to endanger entire species. Very old shrublands could be allowed to burn under all circumstances, including the most severe.

Because the examples presented in this paper were shrub or tree species, it could be argued that the scheme presented above is too dendro-centric. We could, for example, use controlled fire to mimic Native American burning, which we believe maintained some areas in a more open condition. That is, Native Americans either purposely caused local extinction of woody populations or prevented their reaching full development. Unfortunately, the biota is not the same now as it was before 1769. High frequency burning in coastal southern California does encourage some native species, but it cannot completely exclude undesirable exotics. Further research on this point is needed.

Present management goals for fire regimes tend to be dominated by economic and safety considerations. Demands for the protection of life and property provide pressure for controlled burning to maintain low fuel loads. This can only be accomplished if fire rotation is short (<30 years), even though historical data suggest that shrublands generally burned at 50 or more years. Further, the assumption that "breaking up" uniform fuel beds precludes catastrophic fires is belied both by the historical record, and by the 1993 spate of fires that was the impetus for the conference where this paper was presented.

Recently, managers have also had to deal with limitations imposed by rare and endangered species. In some cases, burns have had to be deferred because of concerns for listed species. This may be a problem for an economically-driven program, but, as explained above, it is of little concern from a biological point of view. Perhaps some habitats need to be burned to maintain rare species, but we have time to do the research necessary to establish the validity of this view.

It would seem, then, that we are at an impasse. A biologically desirable fire regime may not be compatible with the socioeconomic regime. The two could be brought closer together except for one key difficulty:

wildfire destructiveness is worse than it would need to be because of poor landuse planning and unrealistic expectations for fire protection. The general public refuses to accept that catastrophic wildfires are inevitable and that where shrublands meet human development, disaster is a likely result. Californians tend to reject the intrusion of government in the form of landuse planning and building codes, but seem to welcome it as expensive fire protection.

Those of us with a biologically-driven agenda hope that more sensible landuse patterns will be adopted, but history cautions us not to expect it. For the future, the public will probably demand a level of fire protection that is incompatible with what is best for natural ecosystems. It will therefore probably be necessary to utilize a combination of brush reduction and clearance and controlled burning along the urban/wildland interface. There may be no need, however, to impose this socio-economic regime everywhere. Efforts should be made to exempt more remote areas from short-rotation controlled burning and perhaps from controlled burning of any kind. Because of habitat fragmentation, such a policy might cause some areas of shrubland to age beyond the historical average, but, as argued above, the senescence risk appears to be minimal; in any case, we have many decades to refine our knowledge of its magnitude.

The Future

To increase our knowledge we must take every opportunity to answer critical questions about how fire affects the integrity and biodiversity of our natural systems. Because there is always more money for management than for research, it follows that we should utilize management actions as experiments and opportunities to observe the response of the vegetation. This is the central concept of what has been called "adaptive management" (Walters 1986).

In the meantime, the inhabitants of the ivory towers must recognize that managers are under severe pressure to act, and must justify their actions to a broad constituency. The managers in their turn must acknowledge the complexity of the systems they are managing, and the impossibility of being able to predict if a particular research plan will be optimal, or even if it will be able to achieve the stated goals. Of necessity, managers operate on partial information, on the basis of poorly-tested hypotheses, and sometimes on pure speculation. The challenge is to integrate management and research without losing the capacity for critical examination and rigorous testing of ideas which is the only means by which we will advance our knowledge of southern California shrublands.

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