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# CONSERVATION CONFLICTS OVER BURNING BUSH IN SOUTH-EASTERN AUSTRALIA

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**Abstract**

*Current fire management practices in the fire-prone vegetation of south-eastern Australia are based mainly on the concept of hazard reduction via the use of periodic low-intensity fires to maintain the amount of flammable fuel within specified (low) limits. We examined the possible conflict between the requirements of fire management for hazard reduction and requirements for species conservation in the dry-sclerophyll shrublands and woodlands of the Sydney region. Our data indicate that potentially severe fire hazards (fine fuel loads of ≥10 t.ha$^{-1}$) can reappear in the woodland and shrubland vegetation within 2–4 years after low-intensity fires, such as are typical of the fuel-reduction burns usually prescribed. Our data also show that low-intensity fires will have significant effects on the species composition of the communities if they occur with an inter-fire interval of less than 7–8 years, causing a significantly reduced abundance of long-lived woody shrub species.*

*There is thus a clear conflict in south-eastern Australia between fire management practices based solely on prescribed burning for hazard reduction and the fire management practices necessary to maintain ecosystem biodiversity, and this conflict is greatest for fire-sensitive shrub species. The conflict between these two vegetation management objectives cannot be resolved by a simple compromise, as prescribed fires with inter-fire intervals any greater than 4 years will allow a potentially severe fire hazard to exist, while a burning regime with any inter-fire intervals less than 8 years will result in loss of biodiversity. This conflict means that it will probably not be possible to achieve simultaneously both hazard reduction and species conservation for any specified managed area.*


*Keywords:* fire, hazard reduction, fuel load, conservation, biodiversity.

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## INTRODUCTION

Australia contains many fire-prone regions, and fire is an important factor determining the growth, survival and persistence of many plant species in Australia (Gill *et al.*, 1981; Pyne, 1991). The characteristics of the fire regime that are important in determining these patterns are (Gill, 1975): fire frequency (how often the fires occur), fire intensity (how hot the fires get), and season of burning (what times of the year they occur). However, current fire management practices in the fire-prone vegetation of the Sydney region are based mainly on the concept of fuel reduction (Luke & McArthur, 1978; Richmond, 1981; Whelan & Muston, 1991); that is, the fire regimes that are prescribed for a particular managed area are determined principally by the amount of fire fuel that has accumulated in that area. This is clearly an important criterion for fire management, as it reduces the hazards associated with subsequent non-prescribed fires (wildfires) in the managed area (Gill *et al.*, 1981).

The development of hazard-reduction burning as a management technique in Australia was largely a product of the need to manage forests for wood production (Richmond, 1981; Pyne, 1991; Whelan & Muston, 1991), and was thus designed primarily to protect commercial timber resources. This means that these fire-management practices are not based on concepts such as maximizing or even maintaining species conservation (and thus biodiversity), in spite of the fact that these practices will have a significant effect on the continued long-term survival of plants in the managed area (Gill *et al.*, 1981; Whelan & Muston, 1991; Williams *et al.*, 1994). If there is a conflict between the needs of vegetation management for hazard reduction and management for species conservation, then this will have important consequences for areas where management objectives include preservation of biodiversity (Good, 1981; Pratten, 1984; Worboys & Gellie, 1989).

A number of simple demographic models have been developed for plant species in southern Australia,

**Table 1. Summary of the plant characteristics that are important in the demographic models**
(Data are the approximate time in years)

| Fire tolerance | Juvenile period | |
|---|---|---|
| | Primary | Secondary |
| *Herbaceous/semi-woody plants* | | |
| Sensitive | 1–5 | — |
| Tolerant | 4–6 | 1–2 |
| *Woody plants* | | |
| Sensitive | 4–8 | — |
| Tolerant | 7–14 | 2–3 |

which allow us to make predictions about plant community behaviour under particular fire regimes (e.g. Bradstock & Auld, 1987; Bradstock, 1990; Cowling *et al.*, 1990; Burgman & Lamont, 1992). Two plant characteristics seem to be of most importance in relation to fire frequency (Gill, 1977): (1) whether the plants are fire-sensitive (>50% adult mortality of a population subject to 100% leaf scorch) or fire-tolerant (<50% adult mortality of a population subject to 100% leaf scorch); and (2) the length of the primary juvenile period (the time taken for juvenile plants to reach first reproduction for fire-sensitive species, and the time taken by juvenile plants to reach fire-tolerant size for fire-tolerant species) and the length of the secondary juvenile period (the time taken for adults that have survived the fire to reach first post-fire reproduction). So, for example, local extinction of a fire-sensitive species will occur if the fire regime consists of inter-fire intervals that are shorter than the primary juvenile period (e.g. Bradstock & O'Connell, 1988). Similarly, new individuals of a fire-tolerant species will not be successfully recruited to a population if the fire regime consists of inter-fire intervals shorter than either the secondary juvenile period or the primary juvenile period (e.g. Bradstock & Myerscough, 1988).

These demographic models indicate that it is the woody shrubs that are likely to be the most affected by relatively short inter-fire intervals (Table 1). The herbaceous and semi-woody species have comparatively short primary and secondary juvenile periods, allowing them to complete their life-cycle successfully even under a series of short inter-fire intervals, while the tree species are very long-lived and will thus not be noticeably affected until the fire regime occurs over an extremely protracted period. Any conflict between fire management for hazard reduction and for species conservation should thus become apparent first in changes in the abundance of the woody shrubs in the managed area. In particular, those shrub species that have a canopy-stored seedbank should be affected before those with a soil-stored seedbank, as their seedbank will be depleted more rapidly (Bradstock & Auld, 1987). There is a need to test the predictions of long-term community change produced by these simple demographic models (Whelan & Muston, 1991).

As a quantitative test of the potential conflict between the requirements of vegetation management for hazard reduction and for species conservation, we examine these aspects in detail for one of the most common vegetation types of the Mediterranean-type ecosystems of the Sydney region in south-eastern Australia. The sclerophyll forests, woodlands and heaths of this region are prone to periodic large conflagrations during summer if high fuel loads (mostly litter) are allowed to develop, and thus fire management is via deliberate hazard-reduction fires applied relatively frequently in autumn and/or spring (Richmond, 1981; Whelan & Muston, 1991). We quantify those vegetation characteristics that are associated with increasing fire hazard, and then the floristic effects that manipulating these characteristics with prescribed fires will have on the vegetation.

## STUDY AREA AND METHODS

Our work was carried out on the Lambert Peninsula in Ku-ring-gai Chase National Park, a conservation reserve (*c.* 15,000 ha) on the northern outskirts of Sydney. All of our samples were taken from a vegetation mosaic of intergrading closed-scrub/scrub-heath/low woodland/low open-woodland, which occupies 42% of Ku-ring-gai Chase National Park (Thomas & Benson, 1985), and which is the most widespread natural vegetation type remaining in the Sydney region (Beadle, 1981; Benson & Howell, 1990). The shrubland and woodland extremes of this vegetation mosaic can have quite different fuel loads, but their floristic composition is quite similar. All samples were taken from Hawkesbury Sandstone plateaus and ridgetops (*c.* 150 m altitude). The area was almost completely burnt by a high-intensity wildfire in 1965, and since that time much of it has been burnt by a complex mosaic of overlapping low- to moderate-intensity wildfires and prescribed fires. Most of the area was re-burnt by a moderate- to high-intensity wildfire in 1994.

### Hazard reduction

The most important measure of the controllability of a fire is its intensity (measured as kilowatts per metre of fire front) (Luke & McArthur, 1978), and this is thus used as the measure of the hazard associated with a fire. The notional upper limit for fire intensity is about 100,000 $kW.m^{-1}$ (Trevitt, 1994), and extremes of the order of 60,000 $kW.m^{-1}$ occur in forest wildfires in south-eastern Australia (Luke & McArthur, 1978). The upper limit for practical fire control is approximately 3500 $kW.m^{-1}$ (Gill *et al.*, 1987; Trevitt, 1994), and the normal limit for planned (prescribed) fires is about 500 $kW.m^{-1}$ (Luke & McArthur, 1978).

Fire intensity ($I$) may be estimated as the product of the heat yield of the fuel ($H$; kJ per kg), the weight of the combustible fuel ($W$; tonnes per hectare), and the rate of forward spread of the fire front ($R$; km per hour):

$$I \approx H \times W \times R$$

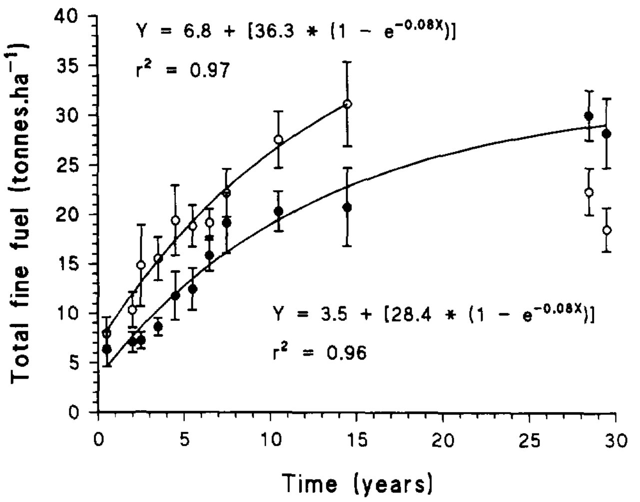


**Fig. 1.** Pattern of accumulation of combustible fine fuel in Ku-ring-gai Chase National Park woodland (○) and shrubland (●) communities through time after low-intensity fires. Each symbol represents a site with a different time since the most recent fire, and the error bars are standard errors from the 10 replicate quadrats at each site.

(Luke & McArthur, 1978; Richmond, 1981; Walker, 1981). The heat yield of the fuel will be determined mainly by the floristic composition of the vegetation (with respect to its organic content and flammability), and its moisture content; therefore, a 'typical' average value can be estimated for any specified area. The weight of the fuel is also largely determined by the floristic composition of the vegetation, and by the rate at which the fine fuel (litter, leaves, and branches <6 mm diameter) accumulates through time with plant regeneration after the previous fire; these parameters are potentially under human control by burning or harvesting. The rate of forward spread of the fire is related to wind speed and direction, the moisture content of the fuel, and topography; and these parameters are uncontrollable in any specific set of circumstances.

Consequently, the intensity of both planned and unplanned fires can only realistically be managed by manipulating the amount of fuel that is available to be consumed. Since the 1960s, hazard reduction has thus normally involved maintaining fuel loads within specified limits by creating periodic low-intensity fires to consume excess (particularly litter) fuel (Luke & McArthur, 1978; Pyne, 1991; Whelan & Muston, 1991; Williams *et al.*, 1994). The decision about when to initiate a prescribed burn, given appropriate weather conditions, can thus be made by estimating the available fuel load. An acceptable level of hazard in the sclerophyll vegetation of southern Australia has been considered to be fine fuel loads of about 8–10 $t.ha^{-1}$ (Hodgson, 1968; Gill *et al.*, 1987; McCaw *et al.*, 1992), although this estimate varies depending on both relief and windspeed, and fuel loads can show considerable small-scale spatial heterogeneity.

Fuel accumulation through time can be modelled using a first-order asymptotic completion (or exponential saturation) model of the form:

$$W = Init + [(Limit - Init) \times (1 - e^{(k \times t)})]$$

where *Init* is the fuel load ($t.ha^{-1}$) immediately after the most recent fire, *Limit* is the steady-state fuel load ($t.ha^{-1}$), *k* is the fuel accumulation rate, and *t* is the time (years) since the most recent fire (modified from Fox *et al.*, 1979; Walker, 1981).

We measured the dynamics of fuel loads after low-intensity fires in Ku-ring-gai Chase National Park by harvesting all of the fine fuel components (litter, leaves, and branches <6 mm diameter) in each of 10 randomly located 0.2 $m^2$ × 4 m cylindrical quadrats (Conroy, 1987) at 12 sites during the springs of 1993 and 1994. A sample of the woodland and shrubland vegetation types was taken at each site, each site having a different time since the most recent fire (see Fig. 1 for times). The fire history of each area was determined from the detailed fire history records maintained by the National Parks & Wildlife Service of NSW. The harvested fuel components were oven-dried to constant weight in the laboratory, and then weighed. The first-order asymptotic completion model was fitted to the mean values for each time using non-linear least-squares curve fitting (Wilkinson, 1991), treating the woodland and shrubland samples separately.

This sampling regime examines long-term patterns using space-for-time substitution to produce a chronosequence (Conroy, 1987; Pickett, 1989); that is, spatially separated sites of different ages are treated as showing the generalized behaviour of a single site through time. This technique assumes that all of the other environmental variables (e.g. soil, microclimate, fire regime, biotic interactions) contribute equally to each of the experimental samples (or, at least, that their effects are randomized across all of the samples). These assumptions are inherent in most long-term studies (Pickett, 1989), but one consequence of this sampling strategy is that the between-time variability of the samples is higher than might have been obtained by following several replicate sites through time.

In our experimental design we also concentrated on sampling as many sites with a different time-since-fire as possible, rather than sampling replicate sites for a smaller set of times. This is the most efficient sampling design for calculating non-linear regressions (Crawley, 1993), which was our primary objective. However, it does prevent us from making inferences about within-time variability among sites, and any sampling design aimed at estimating fuel load at a particular time-since-fire should naturally include replicate sites.

Since post-fire fuel accumulation patterns can be quite different for fires of a higher intensity, we also fitted the fuel accumulation model to short-term data acquired after the 1994 wildfire. Two replicate sites were sampled for each of the woodland and shrubland vegetation types in areas that had been subjected to either moderate-intensity or high-intensity burning. The intensity of burning at each site was estimated from post-fire characteristics such as apparent degree of consumption of stems and flame scorch height. The

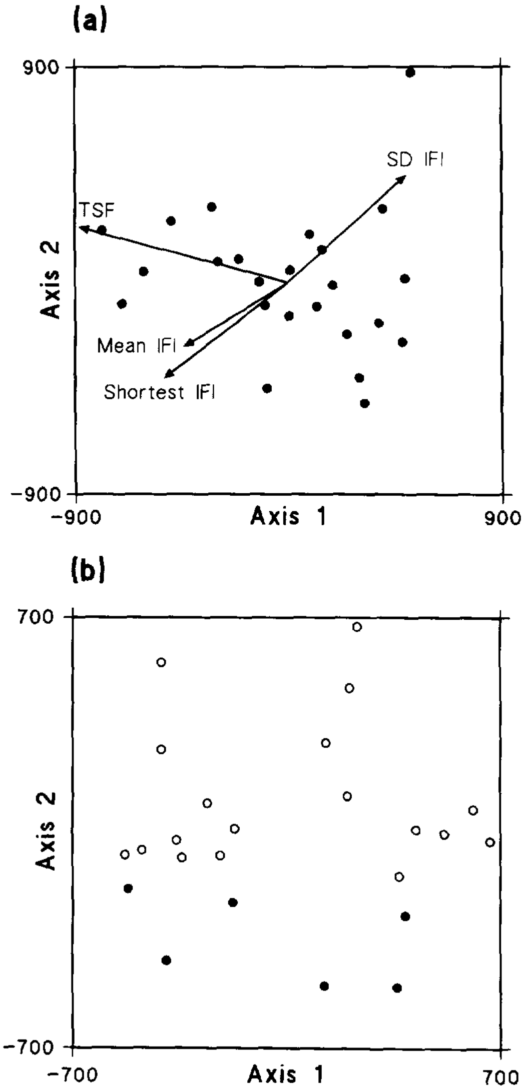


**Fig. 2.** Multivariate analyses of the effects of varying fire frequency on the floristic composition of the shrubland and woodland communities in Ku-ring-gai Chase National Park. Each symbol represents one of the 24 sites sampled, and their spatial proximity indicates how similar they are in terms of their overall vascular plant species composition. (a) Redundancy Analysis, showing the relationship with the components of fire frequency; TSF, time since most recent fire; Mean IFI, mean of inter-fire intervals; SD IFI, standard deviation of inter-fire intervals; Shortest IFI, shortest inter-fire interval. (b) Partial Components Analysis, removing the effect of time-since-fire; those six sites that had been subjected to a shortest inter-fire interval of less than 7 years are highlighted (●).

fuel load at each site was estimated as described above, with five sampling times spread over seven months post-fire.

**Species conservation**

Fire frequency is the number of fires experienced by a particular community within a given time period. The effects of fire frequency on plant communities can be resolved into those resulting from at least two components (Nieuwenhuis, 1987): (1) the time interval between fires; and (2) the time since the most recent fire. These variables are inter-related, since as the number of fires per unit time changes so does the average length of the inter-fire intervals in any one period as well as the time-since-fire. Thus, from a biodiversity perspective hazard-reduction burning actively manipulates time-since-fire, while the effects of the concomitant manipulation of the length of the inter-fire interval are largely unknown (Bradfield, 1981; Fox & Fox, 1986, 1987; Nieuwenhuis, 1987; Bowman *et al.*, 1988; Fensham, 1990; Morrison *et al.*, 1995; Cary & Morrison, 1995).

To examine the floristic effects of the interaction between time-since-fire and inter-fire interval, 24 sample sites were chosen in spring 1990 to represent the range of fire frequencies present on the Lambert Peninsula. Among the sites, time-since-fire varied from 2 to 26 years and inter-fire interval from 1 to 24 years. No distinction was made between woodland and shrubland vegetation, as they are floristically very similar.

Our sampling design assumes that the fire history of each of the sites was similar (or at least randomized) before detailed records were started in the early 1960s. This assumption is reasonable, given that many of the fires in the Sydney region before that time were wildfires that consumed very large areas of bushland with a much lower frequency than occurs today (Kodela & Dodson, 1989), and that the Aborigines and early settlers probably employed a consistent fire management policy for each vegetation type (Clark & McLoughlin, 1986). Our sampling also assumes that the intensity, season and type (i.e. wildfire versus prescribed fire) of fire are randomized across areas. This assumption also appears to be quite reasonable, based on our sampling programme.

For each site, we estimated the abundance of each of the 140 vascular plant species encountered using the importance-score technique of Outhred (1984), each sample consisting of a pair of 100 $m^2$ quadrats. This technique produces abundance scores that are functionally equivalent to frequencies, which are directly related to plant density. The abundance data from the replicate quadrats were pooled for analysis. All species nomenclature follows Beadle *et al.* (1982).

The effects of fire frequency on plant species composition were first analyzed by redundancy analysis (ter Braak, 1988). This is a constrained ordination technique based on principal components analysis that, in a joint analysis of two data sets (e.g. floristic and environmental), assesses the degree to which they show co-variation (ter Braak & Prentice, 1989). That is, it seeks patterns among the samples that occur in both data sets, while ignoring patterns that are unique to either one of the data sets alone. This is thus a direct gradient analysis technique analogous to canonical correlation analysis but which avoids many of the mathematical constraints inherent in that technique (ter Braak & Prentice, 1989), which are unrealistic for most biological data sets.

The redundancy analysis produces two inter-related ordination diagrams, one for each of the two data sets, which can be displayed simultaneously in a species–environment biplot. The degree to which the floristic variation associated with the environmental variables is displayed by the biplot can be assessed by calculating what percentage is shown of the variation shared in common between the two data sets. A Monte Carlo permutation test is used to assess the statistical significance of the association found.

The two data sets analyzed by redundancy analysis were the species abundance (floristic) data and the fire frequency (environmental) data. For the fire frequency data, each of the inter-fire interval times and the time since the most recent fire were $\log_e$ transformed, and the following characteristics were then determined for each sample: time since the most recent fire, the shortest inter-fire interval, the mean inter-fire interval, and the standard deviation of the inter-fire intervals.

The floristic data set was also analyzed by partial components analysis (ter Braak, 1988), which is a modified form of principal components analysis that allows the effects of one or more variables (called co-variables) to be partialled out (removed) by regression techniques prior to the analysis (ter Braak & Prentice, 1989). Time-since-fire was the co-variable, thus allowing us to investigate the effects of inter-fire interval on plant species composition in more detail.

## RESULTS

### Hazard reduction

The fuel accumulation models estimate that 4–7 $t.ha^{-1}$ of fine fuel remains immediately after a low-intensity fire in the study area, and the steady-state fuel load is estimated to be *c.* 32 and 43 $t.ha^{-1}$ for the shrubland and woodland communities respectively (Fig. 1). However, our data also suggest that the fine fuel load in woodland communities never reaches a steady state, the maximum load being 30–35 $t.ha^{-1}$ approximately 20 years after a fire, which is then followed by a decline in fuel load through time (and so our fuel-load model was not fitted to the two oldest samples). This decline is presumably due to senescence and non-replacement of adults of the shorter-lived species, which may then persist in the community as soil-stored seedbanks. The shrubland communities, on the other hand, continue to accumulate fuel for at least 30 years. Note also that we could find no sample areas with a time-since-fire of between 15 and 25 years, due to the historical pattern of prescribed burns and wildfires in the area. As most of the study area was burnt by a wildfire in 1994 (including most of the area with the oldest time-since-fire), there will be no such sites for at least another 10 years; there are also no other large areas in the Sydney region with ages of more than 10 years since the last fire.

The models predict that fine fuel loads of 10 $t.ha^{-1}$ are achieved at about 3.5 and 1.5 years post-fire for the shrubland and woodland communities respectively. It is usually estimated that hazard-reduction burns are actually carried out with an average frequency of about 5 years in the Sydney region (see Discussion), which may allow fuel loads of approximately 13 and 19 $t.ha^{-1}$ to accumulate in the shrubland and woodland communities respectively, producing a severe fire hazard in both vegetation types.

### Species conservation

The redundancy analysis found a statistically significant association between the floristic data and the fire frequency characteristics ($p = 0.050$, from 1000 permutations). The first two axes of the species–environment biplot from the redundancy analysis are shown in Fig. 2(a), these axes accounting for 68% of the total variation of the constrained ordination. The similarities among the samples are indicated by the spatial relationship of the points on the ordination — points near each other show more similarity among themselves (based on their floristic characteristics) than they do to points further away; and the influence of each of the fire frequency characteristics is indicated by the direction and length of the arrows — longer arrows indicate a more influential factor, and orthogonal arrows indicate unrelated factors.

This analysis thus shows a clear distinction between the effects of time-since-fire and the effects of inter-fire interval on floristic composition, as these two characteristics each have a relationship with floristic composition that is about equal in magnitude but different in effect (equal-length but almost orthogonal arrows). This means that variation in these two characteristics is associated with changes in the abundance of different plant species, and that the floristic composition of any one area will thus be a reflection of the combined influence of these two variables. In these analyses there is very little distinction between the effects of variation in the three measures of inter-fire interval (co-linear arrows).

The partial components analysis (removing the effect of time-since-fire) shows that the floristic composition of those sites that had been subjected to at least one inter-fire interval of less than 7 years is distinguishable from the floristic composition of sites that had only experienced longer intervals (Fig. 2(b)). This means that the relative abundance of certain plant species in the community is related to the length of the time between fires, and that the imposition of prescribed burns at intervals of 7 years or less will significantly modify the species composition of the affected communities.

As predicted by the demographic models, the species most affected by this fire regime include two short-lived semi-woody shrubs (*A. suaveolens* and *Z. laevigata*) that increase in abundance under the short-interval regime, and five long-lived woody shrub species with canopy-stored seedbanks (*B. ericifolia, H. sericea, H. teretifolia, L. formosa* and *P. pulchella*) that decrease in abundance (Table 2). These latter shrubs are among the top 10% in terms of abundance in the data set, and they are considered to be dominant or characteristic species for this vegetation type in Ku-ring-gai Chase National Park (Outhred *et al.*, 1985; Thomas & Benson, 1985) and in the Sydney region (Beadle, 1981;

**Table 2. Plant species whose abundance was significantly affected by inter-fire intervals of less than 7 years**

| Species | Family | Fire tolerance | Probability[a] | Average abundance[b] | |
|---|---|---|---|---|---|
| | | | | < 7 years | > 7 years |
| *Acacia suaveolens* | Mimosaceae | Sensitive | 0.031 | 4.3 | 1.1 |
| *Banksia ericifolia* | Proteaceae | Sensitive | 0.002 | 3.5 | 9.2 |
| *Hakea sericea* | Proteaceae | Sensitive | 0.037 | 2.6 | 5.8 |
| *Hakea teretifolia* | Proteaceae | Sensitive | 0.014 | 3.5 | 8.3 |
| *Lambertia formosa* | Proteaceae | Tolerant | 0.041 | 3.6 | 5.5 |
| *Petrophile pulchella* | Proteaceae | Sensitive | 0.038 | 3.7 | 8.6 |
| *Zieria laevigata* | Rutaceae | Sensitive | 0.003 | 1.4 | 0.0 |

[a]From a one-factor analysis of covariance (Wilkinson, 1991) comparing the six sites burnt with an interval of <7 years to the remaining 18 sites (with time-since-fire as the covariable).
[b]Frequency out of 14 sub-quadrats.

Benson & Howell, 1990). Species with lower abundance may also show the same patterns, but our sampling programme was designed to detect overall community patterns rather than patterns in individual species and may thus not have been intensive enough to detect these patterns.

## DISCUSSION

Our data indicate that potentially severe fire hazards (fuel loads of ≥10 t.ha$^{-1}$) can reappear in the sclerophyll woodland and shrubland vegetation of the Sydney region within 2–4 years after low-intensity fires, such as are typical of the hazard-reduction burns often prescribed for vegetation management; and similar results have also been reported for several other sclerophyll forest types in south-eastern Australia (Fox *et al.*, 1979; Raison *et al.*, 1983; Birk & Bridges, 1989). Our data also show that these low-intensity fires will have significant effects on the species composition of the woodland and shrubland communities if they occur with an inter-fire interval of less than 7–8 years. Even one such inter-fire interval can dramatically reduce the abundance of many of the species, and continuation of this fire regime will significantly affect the composition of the community (Cary & Morrison, 1995).

There is thus a clear conflict in the Sydney region between fire management practices based solely on prescribed burning for hazard reduction and the fire management practices necessary to maintain ecosystem biodiversity. This conflict is greatest for fire-sensitive shrub species whose adults are usually killed by fire (and that have a canopy-stored seedbank), as local extinction of a population may occur if an inter-fire interval is shorter than the time taken for the plants to reach first reproduction, and this can be greater than 7 years for many woody shrubs. Faster-growing herb and grass species will not be as greatly affected by short inter-fire intervals, and long-lived tree species will only be noticeably affected over several decades or even centuries. However, the conflict also exists for woody fire-tolerant shrub species (such as *L. formosa* in our study), as local extinction may occur if the fire regime consists of inter-fire intervals that are shorter than the time required for a species to become fire-tolerant, so that new individuals can be recruited to the population.

The conflict between these two vegetation management objectives cannot be resolved by a simple compromise. Prescribed fires with inter-fire intervals any greater than 4 years will allow a potentially severe fire hazard to exist, while a burning regime with any inter-fire intervals less than 8 years will result in loss of biodiversity. The simple compromise fire frequency, utilizing inter-fire intervals of 4–8 years, will not achieve either hazard reduction or species conservation, and thus will not produce a balance between the two.

This absolute conflict between the two vegetation management practices means that it will probably not be possible to achieve simultaneously both hazard reduction and species conservation for any specified managed area, and that land managers will therefore have to set very clear objectives to determine which management practice is most appropriate for each area (Pratten, 1984; Underwood, 1989; Worboys & Gellie, 1989). This is particularly true for conservation areas, where management objectives must include preservation of biodiversity but may also include the conflicting requirements of hazard reduction. For example, in the Sydney region the *National Parks & Wildlife Conservation Act 1975* requires 'the preservation of the park or reserve in its natural condition' while the *Bushfires Act 1949* requires managers to 'take all practicable steps to prevent the occurrence of fire and to minimise the danger of the spread of fires' (Worboys & Gellie, 1989; Trevitt, 1994). Furthermore, the *Environmental Planning & Assessment Act 1979* necessitates that both of these requirements be considered, by insisting that any fire management activity be assessed in relation to its potential impact (Worboys & Gellie, 1989; Trevitt, 1994), and the *National Forest Policy Statement 1992* recognizes that prescribed burning can be used for management programmes related to either hazard reduction or species conservation (Williams *et al.*, 1994). Our data indicate that these multiple legislative requirements will not be easy to reconcile.

South-eastern Australia has a number of characteristics that make it difficult to develop fire-management policies in the face of this conflict, including the juxtaposi-

**Table 3. Predicted post-fire fuel characteristics for fires of different intensity**

| Fuel characteristic | Fire intensity | | |
|---|---|---|---|
| | Low[a] | Moderate[b] | High[c] |
| *Unburnt fine fuel ($t.ha^{-1}$)* | | | |
| Shrubland | 3.5 | 1.0 | 0.2 |
| Woodland | 6.8 | 3.0 | 0.1 |
| *Fine fuel after 5 years ($t.ha^{-1}$)* | | | |
| Shrubland | 12.9 | 7.8 | 5.8 |
| Woodland | 18.8 | 11.4 | 7.6 |
| *Time to reach 10 $t.ha^{-1}$ (years)* | | | |
| Shrubland | 3.2 | 6.9 | 9.6 |
| Woodland | 1.2 | 4.0 | 6.9 |

[a]Values estimated from the equations shown in Fig. 1.
[b]Values estimated from:
shrubland: $y = 1.0 + [30.0 * (1 - e^{-0.05x})]$, $r^2 = 0.45$;
woodland: $y = 3.0 + [30.1 * (1 - e^{-0.07x})]$, $r^2 = 0.48$.
[c]Values estimated from:
shrubland: $y = 0.2 + [30.0 * (1 - e^{-0.04x})]$, $r^2 = 0.38$;
woodland: $y = 0.1 + [29.9 * (1 - e^{-0.06x})]$, $r^2 = 0.53$.

tion of large urban areas with large precincts of natural vegetation (mainly national parks, recreation reserves, and water catchments) (Whelan & Muston, 1991), as well as spatial heterogeneity that makes it difficult for management practices derived for one area to be applied to different areas (Williams *et al.*, 1994), and the fact that decisions regarding the timing of prescribed burning are often made by the local bushfire brigade rather than by the land managers (Whelan & Muston, 1991). It is usually estimated that hazard reduction burns are actually carried out with an average frequency of about 5 years in the Sydney region (Walker, 1981; Benson, 1985; Pyne, 1991; Whelan & Muston, 1991), and land management in south-eastern Australia has thus apparently been based in practice on attempts to compromise between the two conflicting consequences of the vegetation-management practices. This can no longer be considered to be appropriate.

Land managers have also tended to produce generalized management plans for large areas (e.g. whole conservation reserves) that do not necessarily take into account these different consequences (Underwood, 1989). For example, prescribed burning may be used to create a mosaic effect, where fires of variable intensity and patchy distribution at different inter-fire intervals are used (Christensen & Kimber, 1975; Dickinson & Kirkpatrick, 1987), resulting in management areas that do not have a single broad-scale fire regime but where the regime varies with the spatial scale. However, it is not clear that this procedure effectively resolves the conflict that we have identified, as this practice does not necessarily reduce the hazard of a high-intensity wildfire starting in one area and spreading into an adjacent one, nor does it ensure species conservation at any specified location.

The form of conflict discussed here is not a new land management issue, as it has been recognized in urban planning for many years. For example, there is an absolute conflict in the central business district of metropolitan areas between the needs of open space and the existence of high-rise buildings, and this conflict is not successfully resolved by a simple compromise involving the erection of medium-density housing on all of the affected areas. In these cases, different specified areas are managed for distinctly different purposes, and all conflicting management practices are excluded from each of these areas.

A similar overall strategy may be necessary for vegetation management, where some areas (such as, for example, small urban bushland areas or those in close proximity to the urban fringe) are managed solely for hazard reduction, perhaps using methods other than prescribed burning (Pratten, 1984), while other areas are managed solely for species conservation. There will thus be a price to pay in each area, with potential loss of biodiversity in one area and a potentially severe fire hazard in the other, but these costs may have to be accepted (Pratten, 1984). Such a strategy may not be easy to implement, as it implies the existence of sharp boundaries between managed areas, and such boundaries are not often found in nature (although they are easily created in the urban environment).

The amount of available post-fire fine fuel identified for the vegetation types in our study is relatively large and the fuel-accumulation rates are relatively high. This is because the post-fire fuel is not necessarily composed only of available pre-fire fine fuel that remained unconsumed by the fire. A low-intensity fire can also potentially increase the amount of post-fire fuel by, for example, converting non-fine fuel into fine fuel (e.g. branches >6 mm diameter being only partially consumed so that they become branches <6 mm diameter), or converting non-available fuel into available fuel (e.g. leaves above the flame height being scorched but not consumed so that they subsequently fall and become leaf litter). The extent to which this counteraction between fuel consumption and fuel creation occurs will be determined mainly by the intensity of the fire (as well as by the vegetation structure), and this affects the effectiveness of hazard-reduction burning.

Our long-term fuel accumulation data, and thus our discussion, are based only on fires that burnt with a low or low–moderate intensity, and the post-fire accumulation patterns will be quite different for fires of a higher intensity (Birk & Bridges, 1989). Fitting the fuel accumulation model to our short-term post-fire data (Table 3) shows that fires with a higher intensity leave much less available post-fire fuel, and that the post-fire fine fuel accumulation is also much slower, a result also found by Birk and Bridges (1989) for an area north of Sydney. This suggests that fires of higher intensity may be more appropriate for hazard reduction in the vegetation types examined here, as they could be applied at a lower frequency and thus have less effect on biodiversity. However, even with these fires, potentially severe fire hazards (fuel loads of 10 $t.ha^{-1}$) can still reappear within the primary juvenile period of the woody fire-sensitive shrubs (Table 3).

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