# BIOLOGICAL INDICATORS OF APPROPRIATE FIRE REGIMES IN SOUTHWEST AUSTRALIAN ECOSYSTEMS

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## ABSTRACT

In the fire-prone natural ecosystems of southwest Australia, modern fire management aims at minimizing the adverse impacts of wildfires on human life, property, and biodiversity. Regular low-intensity prescribed burning to manage fuel build up is a crucial element of wildfire control in many of these ecosystems, particularly eucalypt forests. While the fire protection benefits of this practice are well recognized, the practice is controversial, with opponents claiming that fuel reduction burning regimes lead to long-term ecological damage.

In the absence of long-term data, studies of the regeneration responses and life history attributes of key flora and faunal elements can be used to provide baseline information for developing fire regimes appropriate to the maintenance of biodiversity. Together with knowledge of fuel accumulation rates and fire behavior, this information is being used to design prescribed fire regimes which will achieve both conservation and wildfire protection objectives. Key elements of these regimes are fire frequency, season of burning, fire intensity, patchiness, and scale of burning.

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## INTRODUCTION

Accumulations of flammable vegetation and seasonal hot, dry weather have ensured that fire is an important environmental factor which has shaped forest, woodland and heathland ecosystems of southwest Western Australia. The unique and often rich assemblages of flora and fauna display a diverse array of physical and behavioral adaptations which have enabled them to persist in this fire-prone environment (Gardner 1957, Christensen and Kimber 1975, Burrows et al. 1995). Aborigines had an extensive and practical understanding of the role of fire as a regenerative force and prior to European settlement, they used fire extensively for myriad purposes (e.g., see Hallam 1975). To the early European settlers, however, fire was an anathema, a destructive agent to be excluded and suppressed. During the first half of this century, the policy of fire and land management agencies was largely one of fire exclusion and suppression, with relatively small-scale strategic strip burning to protect young regenerating forests following cutting (see McCaw and Burrows 1989, Burrows 1994). However, this policy was reviewed in the 1950's following a spate of large, intense, and damaging forest wildfires. Fire suppression was not always possible and a regime of large, intense wildfires was neither socially nor ecologically acceptable. Greater emphasis was placed on broad area management of fuel buildup as a means of controlling wildfires and since the 1960's, up to 240,000 hectares of forest are prescribe burned annually by low-intensity (<350 kilowatts per square meter) fires set under cool, moist conditions in spring or autumn.

The frequency with which ecosystems are prescribe burned for fuel reduction depends on the rate of fuel accumulation and on the values at risk from damage by wildfire. Fuel accumulation models have been developed for the major vegetation types and the threat posed by wildfire to specific land units is systematically evaluated by a Wildfire Threat Analysis (Muller 1993). The quantity of fuel available for combustion in the flaming zone largely determines the damage potential and suppression difficulty of fires burning under hot, dry weather conditions. For jarrah (*Eucalyptus marginata*) forests, the fuel management objective is to maintain fuel loads below about 8 tonnes per hectare across most (but not all) of the forest. This is achieved by prescribed burning every 6–10 years, depending on the rate of fuel accumulation which is determined by rainfall and forest canopy cover (see Burrows 1994). For the wetter karri (*Eucalyptus diversicolor*) forests, it is acceptable for fuel loads to accumulate in excess of 15 tonnes per hectare because of the lower risk of wildfires due to the more mesic environment.

The responsible use of fire as a land management tool requires a firm understanding of fire behavior and

Table 1. General description of fire ecology research sites in eucalypt forests of southwest Australia.

| | McCorkill forest | Lindsay forest | Yendicup forest |
|---|---|---|---|
| Mean annual rainfall (millimeters) | 1,050 | 1,200 | 750 |
| Soil type | Yellow/grey gravelly sand | Brown gravelly loam | Yellow/grey gravelly sand |
| Landform | Ridge | Midslope | Ridge |
| Overstory species | *Eucalyptus marginata, E. calophylla* | *E. marginata, E. calophylla, E. diversicolor* | *E. marginata, E. calophylla* |
| Forest height (m) | 20–25 | 30–35 | 15–20 |
| Number of vascular plant species recorded | 222 | 93 | 132 |
| Percent cover and height of mature understory | 60%, 1.0–1.5 m | 65%, 1.5–2.0 m | 25%, 0.5–1.0 m |

of the ecological effects of prescribed fire regimes. While the wildfire protection benefits of rotational fuel reduction burning are well recognized (Underwood et al. 1985), critics of this practice claim that such a regime is ecologically damaging, although there is no scientific evidence to support this claim (Christensen and Abbott 1989). The scientific literature on the short- and medium-term effects of fire is substantial, but there are few published data on the long-term effects of fire regimes. Studies are currently under way in the southwest forests to investigate long-term effects but it is likely to be some time before these studies yield conclusive results. In the absence of long-term data, we propose that the likely fire response patterns of flora and fauna can be anticipated by understanding postfire regeneration strategies and life histories of floral and faunal assemblages (Friend 1993, Whelan 1995). This paper reports on some findings of such studies and discusses how this knowledge can be applied to develop appropriate fire regimes which are consistent with protection and conservation objectives in major southwest Australian ecosystems.

## METHODS

### Postfire Regeneration and Juvenile Period of Plants

This study was part of a broader investigation into the long-term effects of various fire regimes on the floristics and structure of upland jarrah forests (including a fire exclusion treatment). Three study sites were selected to represent upland vegetation across a range of rainfall zones. A brief description of these sites is contained in Table 1. Following postfire observations, all species on the study sites were classified according to Gill's (1981) regeneration response classes (Table 2). Postfire flowering inspections were made at 3–4 weekly intervals until all species had flowered. For each species, the proportion of the population in flower and the relative abundance of flowers on individual plants were estimated at each inspection. A species was deemed to have reached flowering age when more than 50% of the population had flowered.

Table 2. Proportion of plant species by postfire regeneration strategies for three forest sites in southwest Australia.

| Site | Resprouters (soil suckers, lignotuber, rhizome, basal sprouts, large apical bud, geophytes, epicormic shoots, etc.) | Seeders (canopy and soil stored seed) |
|---|---|---|
| McCorkill forest | 70.2% | 29.8% |
| Lindsay forest | 72.3% | 27.7% |
| Yendicup forest | 75.0% | 25.0% |

### Postfire Seedling Density and Floristics

The effect of season of burn on seedling density and floristic composition was assessed at the McCorkhill location from plots which were burned by low-intensity (<350 kilowatts per square meter) fire in spring and in autumn, and in plots which had not been burned for 18 years (long unburned control). Two plots each of 4 hectares were assessed for each of the three treatments. A total of 20 × 2 meter$^2$ quadrats were located in each of the treatments (10 in each plot) to assess seedling regeneration 12 months after fire. The number of seedlings of each species was recorded for each quadrat. Without mature foliage or flowers, it was sometimes difficult to taxonomically identify seedlings. Where this was the case, seedlings were given a unique numeric code.

### Faunal Responses to Fire

Most studies of the impact of fire on fauna have been short term. They either assess the postfire response of fauna or describe population changes since fire occurred. The various faunal response patterns to fire reported here were obtained during intensive studies with replicated treatments and controls in a variety of southwest Australian vegetation types. Data on population dynamics and activity patterns of specific species in a variety of habitats were gathered by regular trapping before and after fire and by radio telemetry.

In the wheatbelt woodlands and shrublands vegetation types, two mammal species with different life histories were studied during a small-scale (100 hectares) experimental autumn burn carried out at Tutanning Nature Reserve. The red-tailed phascogale (*Phascogale calura* Marsupialia: Dasyuridae), once widely distributed across central and southern Australia, is now restricted to remnant vegetation in parts of the wheatbelt, where the average annual rainfall ranges from 300–600 millimeters (Kitchener 1981). This small (40–60 grams) carnivorous marsupial occurs in flammable wandoo (*Eucalyptus wandoo, E. accedens*) and rock sheoak (*Allocasuarina huegeliana*) communities. The species is arboreal, utilizing tree hollows for refuge and for nesting sites. It has a highly synchronous breeding system during late winter, after which all males die (Kitchener 1981, Bradley 1987). The response of the little long-tailed dunnart (*Sminthopsis dolichura* Marsupialia: Dasyuridae) to fire was also studied in this vegetation type. This animal is a small (20–30 grams) mouse-like, insectivorous marsupial, which, unlike the red-tailed phascogale, is ground-dwelling, sheltering in low shrubs, under bark or in hollows in logs. It has a high reproductive rate with females capable of raising more than one litter per year. Males do not die after breeding (Friend and Pearson 1995).

Two small species were also studied in detail in the south coast mallee-heath vegetation type. The first of these, the honey possum (*Tarsipes rostratus* Marsupialia: Tarsipedidae) is a unique marsupial with a highly specialized diet of pollen and nectar which it gleans from the flowers of the floristically rich heathlands which characterize much of south coastal areas of Western Australia. This small (10–20 grams) animal breeds continuously and its high metabolic rate requires it to take in food on a daily basis (Renfree et al. 1984). The second animal studied in this vegetation type was the ash-grey mouse (*Pseudomys albocinereus* Rodentia: Muridae), also a small (15–40 grams) animal which shelters in burrows in sandy soil. It is a generalist in its diet, eating seeds, grasses, herbs, and some invertebrates. It breeds opportunistically, depending on the availability of resources, and under good conditions has a considerable reproductive capacity (Morris 1995).

Other groups of animals studied and reported on here include frogs, reptiles, and invertebrates. The numerical response patterns of these groups to fire was measured by pitfall trapping before and after fire and in adjacent control (long unburned) sites.

## RESULTS AND DISCUSSION

### Response of Jarrah Forest Flora to Fire

Of about 300 jarrah forest understory species examined across three rainfall zones, 70–75% of species resprouted following fire, either from epicormic buds buried beneath protective bark, apical buds or subterranean organs. The remainder depended on seed stored in the canopy or in the soil (Table 2). The proportion of resprouters-to-seeders is consistent with other observations for southwest forest ecosystems (Christensen and Kimber 1975, Burrows et al. 1987).

Time to first flowering after fire (juvenile period) is an important parameter for determining the minimum fire interval to ensure persistence of obligate seed species, especially those which depend on seed stored in the canopy for regeneration. The juvenile period depended on species, season of fire, season of flowering and on rainfall and was largely independent

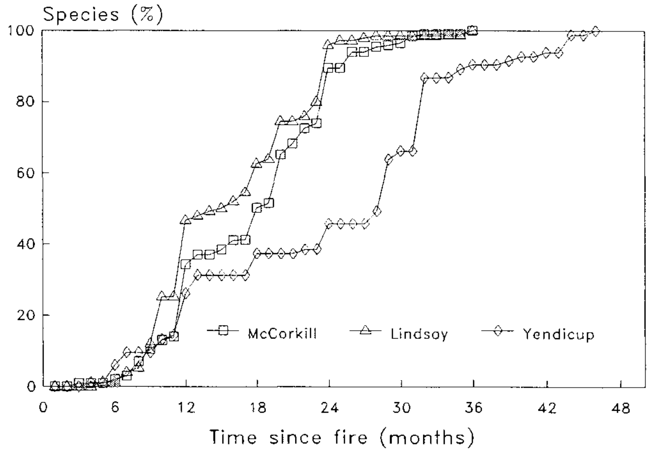


Fig. 1. Cumulative proportion of about 300 understory species to have reached flowering age with time after fire at three sites in the jarrah forest.

of regeneration strategy, although resprouters tended to flower sooner than seeders. In intermediate and high rainfall forests (1,000–1,200 millimeters per year) studied here, about 90% of all recorded understory species flowered within 2 years of fire, with 100% flowering within 3 years. In lower rainfall forests (about 750 millimeters per year), all understory species flowered within 4 years of fire (Figure 1). In the drier forests, the juvenile period was 12–18 months longer for the same species in higher rainfall forests. Within species variability of juvenile period associated with site conditions has also been reported for species in eastern Australia (e.g., Whelan 1995 citing Muston 1987). Mature overstory tree species (*Eucalyptus marginata* and *E. calophylla*) took 4–6 years to recover their crowns and to flower following crown damaging fires (Burrows, personal observation). Unlike summer wildfires, low-intensity (<350 kilowatts per square meter) prescribed fires rarely cause crown damage to mature trees. Understory flowering occurred throughout the year, but as expected there is a seasonal peak with about 75% of species flowering in spring.

Season of flowering and season of fire affected the juvenile period of species which flowered within 12 months of fire. For example, at one site, *Baeckia camphorosmae* flowered in February, 6 months after a spring fire, but at a nearby site plants first flowered in February, 11 months after an autumn fire. On the other hand, some species such as *Burnettia nigricans* flowered in September, 12 months after a spring fire but at an adjacent site the plants flowered in September, 6 months after an autumn fire.

The juvenile period, however, does not necessarily indicate sufficient time for seed set and replenishment of seed banks (Abbott 1985, Benson 1985, Bradstock and O'Connell 1988). While there is limited information on the time for adequate accumulation of seed stores for jarrah forest species, Gill and Nicholls (1989) reported that a conservative fire-free period of about twice the juvenile period was necessary to replenish seed banks. Recently, Kelly and Coates (1995) have reported similar time periods for a number of

Table 3. Definition of fire frequency for various jarrah forest habitats based on the juvenile period of the slowest maturing understory plant species (longest juvenile period = LJP). The fire frequency ratio (FFR) = fire interval (FI):LJP.

| Habitat type | Longest juvenile period (LJP) (yrs) | Sustainable fire interval (2×LJP) (yrs) | FFR <2 High fire frequency (yrs) | FFR 2–4 Moderate fire frequency (yrs) | FFR 4–6 Low fire frequency (yrs) | FFR >6 Very low fire frequency (yrs) |
|---|---|---|---|---|---|---|
| High rainfall (>900 mm) upland jarrah forest | 3 | 6 | <6 | 6–12 | 12–18 | >18 |
| Low rainfall (<900 mm) upland jarrah forest | 4 | 8 | <8 | 8–16 | 16–24 | >24 |
| High rainfall (>900 mm) riparian jarrah forest | 6 | 12 | <12 | 12–24 | 24–36 | >36 |

*Banksia* species endemic to southwest Western Australia. Adopting this rule-of-thumb, a fire-free period of about 6 years for intermediate and high rainfall jarrah forests and about 8 years for lower rainfall forests is deemed adequate for all understory species on upland sites to replenish seed banks. These intervals are viewed as conservative minimum fire-free periods; it is highly likely that the seed store is sufficient to ensure adequate regeneration following occasional fires at shorter intervals. However, a sustained regime of fire at frequencies less than those suggested above is likely to result in local decline of some obligate seed species, especially those which depend on seed stored in the canopy. Such "fire sensitive" taxa in the jarrah forest are most likely to be found in habitats which are less fire prone because they either stay moist for a longer period than the upland sites, such as riparian zones, or they carry light or patchy surface fuels (treeless moisture gaining sites) and only burn under warm, dry and windy conditions. For example, the obligate seed species *Lambertia rariflora* is restricted to creek lines in parts of the jarrah forest and has a juvenile period of 5–6 years (Burrows, personal observation). These habitats usually do not burn during fuel reduction burns in spring because the fuels are too moist.

As discussed, the frequency with which strategically important forests are prescribe burned to manage fuel levels depends on the rate of accumulation of fine dead fuels (leaves, twigs, and bark) which accumulate

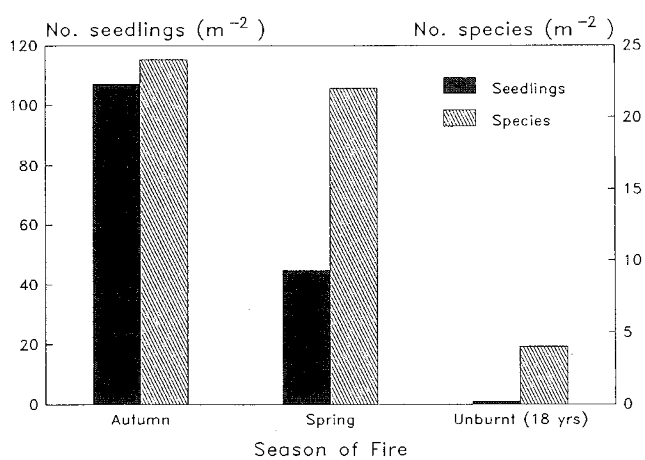


Fig. 2. Jarrah forest seedling density and species richness measured 12 months after autumn and spring fires and at an adjacent site which was last burned 18 years ago.

on the forest floor and which form the dominant fuel on upland sites. Rate of fuel accumulation is a function of rainfall, canopy cover, and time since fire (Burrows 1994). For the first 10–15 years after fire, fuels in intermediate and high rainfall jarrah forests accumulate at a rate of about 1–1.5 tonnes per hectare so these forests are prescribe burned every 6–7 years. In low rainfall forests, initial fuel accumulation rate is about 0.5–1.0 tonnes per hectare so these forests are prescribe burned every 8–10 years. Based on the juvenile periods and the period required for seed bank replenishment, these fire intervals are compatible with the seed bank dynamics requirements of understory species. This probably explains why there have been no reported or observed declines or extinctions of flora attributable to this regime.

Clearly, fire frequency is important in determining the likely impact of a fire regime on floristic composition. Fire frequency is usually described qualitatively using terms such as "high," "low," "frequent" or "infrequent," or quantitatively by the number of years or months between fires. Without qualification, neither of these measures aptly characterizes fire interval in a biologically meaningful way. We propose that the time to first flowering of the slowest maturing species, or the longest juvenile period (LJP) in a definable fuel or habitat type could be used as a biological standard for defining fire frequency. We define the fire frequency ratio (FFR) as the ratio of the fire interval (FI) and the longest juvenile period (LJP). Examples of biological characterization of fire frequency for several jarrah forests habitat types are contained in Table 3.

The season of fire strongly influenced seedling density in jarrah forests but had little affect on floristic composition, measured 12 months after fire (Figure 2). Compared with spring fires, fires under dry soil conditions in summer/autumn resulted in a higher level of seedling germination and survival in the first year, probably due to the favorable postfire conditions for establishment with the onset of winter rains. Fire ephemerals and herbs comprised most postfire seedling germination. Tree species and "hard seed" understory species such as legumes were favored by dry soil fires. Few seedlings germinated in the absence of fire.

## Responses of Fauna to Fire

The experimental autumn fire carried out at Tutanning Nature Reserve directly killed three out of ten

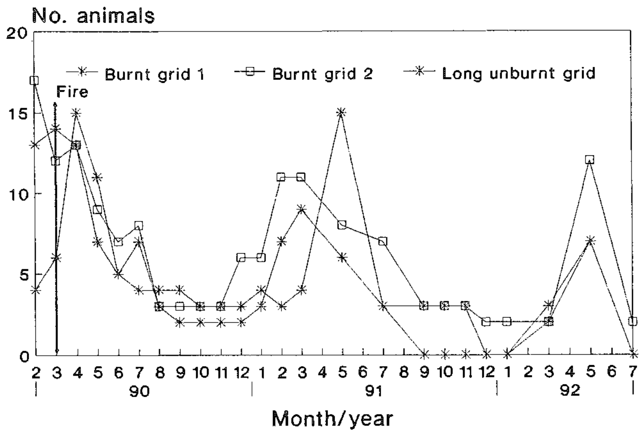


Fig. 3. Number of red-tailed phascogales known to be alive before and after an experimental autumn fire, and at a long unburned (control) site, in Wheatbelt woodland and heath vegetation.

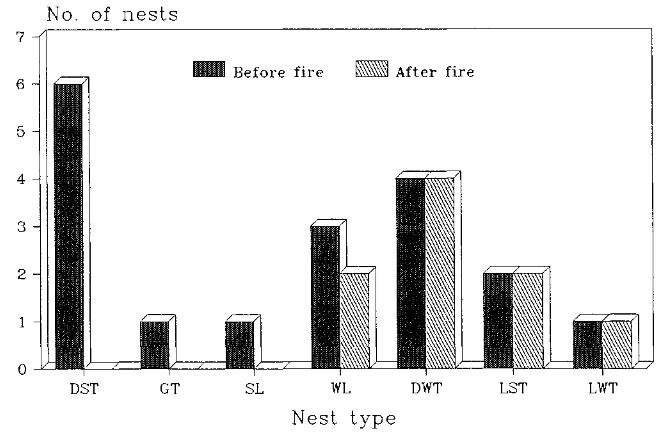


Fig. 4. Nest types used by red-tailed phascogales before and after an experimental autumn fire. DST = dead sheoak tree, GT = grasstree, SL = sheoak log, WL = wandoo log, DWT = dead wandoo tree, LST = live sheoak tree, LWT = live wandoo tree.

radio collared red-tailed phascogales. These animals were killed during the flaming combustion stage of the fire while taking refuge in hollows in rock sheoak trees. The fire intensity was highly variable, reflecting the variable nature of the vegetation (fuel), but ranged from about 200–300 kilowatts per square meter in wandoo woodlands, 700–1,000 kilowatts per square meter in rock sheoak thickets, and 4,000–6,000 kilowatts per square meter in heath (McCaw, personal communication). Typical of fires burning in relatively dry autumn conditions, fuel consumption was virtually complete, especially in the flammable sheoak thickets and heaths which were totally defoliated by fire.

However, as shown in Figure 3, the rate of recolonization of the burned areas by animals from surrounding unburned areas was rapid and within several months, the numbers of animals known to be alive (KTBA) in burned and unburned trapping grid sites was not significantly different. This can largely be attributed to the relatively small size of the experimental burn and the mobility of the animals. Thus, fire scale, and particularly the size of the fire in relation to the size and mobility of the animal, is an important determinant of the likely response pattern of the species to fire. The strong seasonal fluctuations of animal numbers shown in Figure 3 reflect the male die-off following breeding (July) and subsequent recruitment of young animals into the population (November/December).

Another important response to fire displayed by the red-tailed phascogales was the rapidity with which they changed their nest site selection from rock sheoak trees and grass trees (which were consumed or killed by the fire) to wandoo trees, which were better able to withstand the fire and suffered less direct physical impact because of the lower fire intensity (Figure 4, Friend, personal observation). This is a clear demonstration of the flexibility and adaptability of native fauna to disturbances such as fire. The behavioral adaptation of the red-tailed phascogales to quickly adjust and to occupy a new postdisturbance niche is an important ecological observation. However, this change was not without additional risk to the animals. To reach feeding grounds from their new nests in wandoo trees, they had to travel across the ground, predisposing themselves to predation. Long-term baiting of areas to control introduced foxes (*Vulpes vulpes*) and feral cats (*Felis catus*), as was done at Tutanning, may have an influence on the postfire responses of small vertebrates (Christensen and Kimber 1975, Christensen 1978, Friend 1993).

Initially, the rock sheoak thicket regenerated profusely after the fire from seed stored in the canopy. However, 9 months after the fire, the region experienced a serious locust plague which virtually destroyed the regeneration, permanently changing this site from a rock sheoak thicket capable of eventually supporting red-tailed phascogales to a grassland affording little habitat for the species. This is an excellent example of how the scale and timing of disturbance (fire) may interact with episodic events such as locust plagues and antecedent drought to produce a net impact on a species which is largely unpredictable.

In the same vegetation type, the little long-tailed dunnart showed a different response to the same fire. In the first 2 months or so after the fire, the population remained steady, but then fell significantly (Figure 5), as did the population in the control (long unburned) plots. For the first 18 months or so after fire, the populations in both burned and long unburned plots followed similar trends, although the population in the burned plots remained consistently lower than those in the long unburned plots. However, by about 24 months after fire there was no significant difference between the two sites. Unlike the red-tailed phascogales, the increase in the little long-tailed dunnart population following fire was due to both dispersal and *in situ* breeding by animals which survived the fire. Their nonarboreal habits, high reproductive capacity and more generalized food and dietary requirements enable these animals to persist and quickly respond to the conditions prevailing in the early postfire stages. Similar findings have been reported for other species (Fox 1982, 1990, Fox and Whitford 1982, Friend 1993).

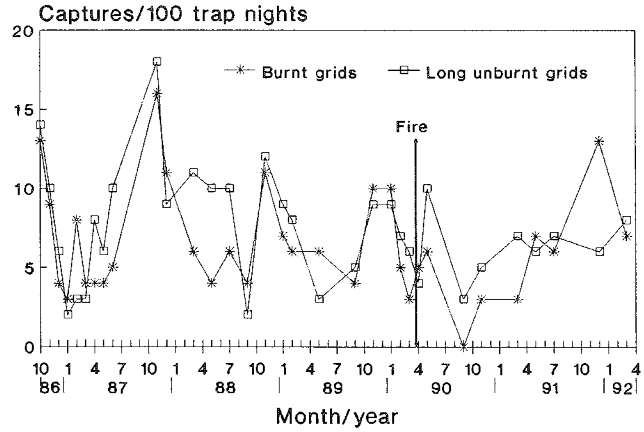


Fig. 5. Numbers of little Long-tailed dunnarts trapped before and after an experimental autumn fire and in a long unburned (control) site in Wheatbelt woodland and heath vegetation.

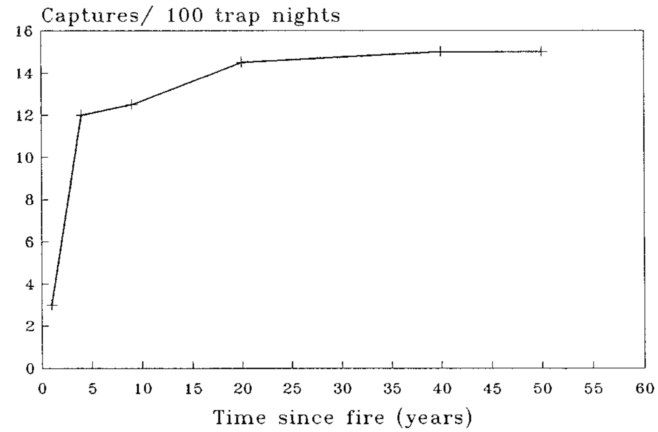


Fig. 6. Capture rate of honey possums with time since last fire in south coast mallee-heath vegetation.

Honey possums are severely affected by large scale high-intensity fires which result in complete or near complete landscape burn-out (Richardson and Wooller 1991). The animals, which usually nest in the elevated vegetation, are either killed by the intense fire, or if they manage to survive the flames, they are likely to be predated or starve within a couple of days if they cannot access suitable flowers.

However, if fires are low intensity, relatively small or patchy, these animals can survive in the landscape and fairly rapid recolonization is facilitated. They will begin to reinvade the burned areas from unburned pockets as soon as a sufficient proportion of the regenerating plants produce pollen and nectar, which may be within 12–18 months of fire. In areas burned by large intense fires, honey possum population density was strongly related to time since fire, or age of vegetation (Figure 6), with density increasing quickly in the first 5 years after fire and reaching a maximum at about 15–20 years after fire. As shown in Figure 6, the relationship between honey possum density and time since fire is nonlinear. About 80% of the potential maximum density of animals is reached within the first 5 years of fire. The time taken to reach maximum density (10–15 years) correlates well with the time it takes for all plants in south coast mallee-heath vegetation types to reach flowering age (McCaw, personal communication). The relationship between honey possum density and time since fire (Figure 6) is similar to the relationship between the number of plants which have flowered and time since fire, suggesting that temporal and spatial availability of nectar and pollen is an important factor controlling site carrying capacity. The season of fire probably has little impact on the honey possum which can breed throughout the year and is relatively sedentary (Wooller et al. 1981).

The ash-grey mouse occurs in similar vegetation as the honey possum, but has vastly different life history and habitat requirements, and displays a very different fire response pattern. Unlike the honey possum, this animal is a disturbance opportunist, with population density highest in the first few years after fire (Figure 7). The ash-grey mouse is able to take advantage of the abundance of herbs, grasses, and seed which are available in the early postfire period. At this time, the vegetation is open, allowing easy access and mobility for this largely ground foraging animal. As the vegetation density increases and as herbs and grasses give way to woody shrubs, the population density of the ash-grey mouse decreases. This pattern is typical of other small native rodents and also of the introduced house mouse (*Mus domesticus*) (Recher and Christensen 1981, Fox 1982, 1990, Friend 1993). However, the native bush rat (*Rattus fuscipes*), a larger rodent which inhabits thick vegetation in creeks and valleys, declined markedly in the first 2–3 years after fire in karri forests (Christensen and Kimber 1975).

Frogs (e.g., *Neobatrachus pelobatoides*) showed little relationship with fire (Figure 7) and, as to be expected, are mostly influenced by seasonal moisture regimes. They mostly burrow deep in the soil during dry periods when fires occur, and therefore evade the acute impacts. Having a generalist invertebrate diet, they are little affected by fire (Bamford 1986, 1992). Some species which are tree dwellers (e.g., *Litoria cyclorhynchus*) are likely to be directly affected by intense fires burning under dry conditions, but there are

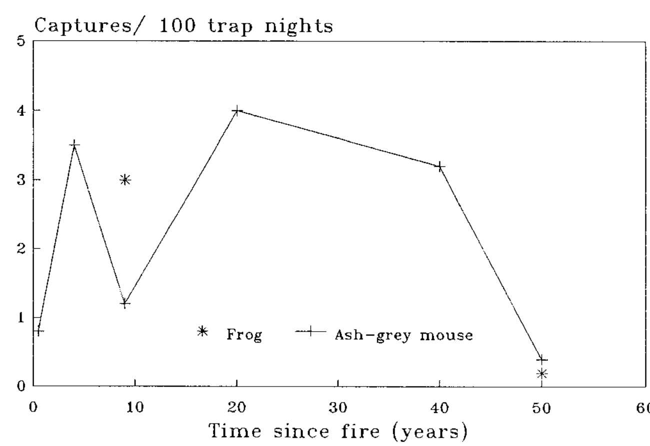


Fig. 7. Abundances of the ash-grey mouse and of the frog *Neobatrachus pelobatoides* with time since fire in south coast mallee-heath vegetation.

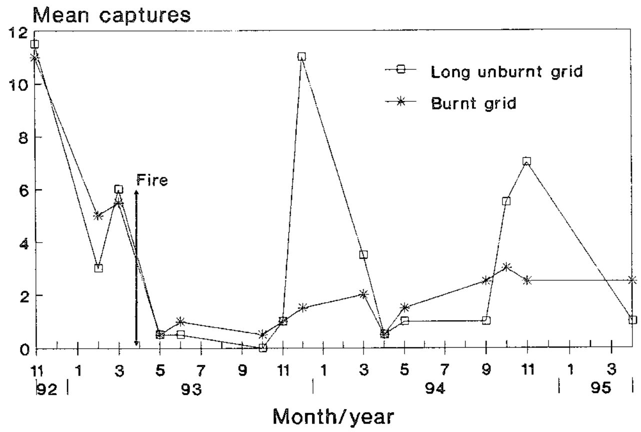


Fig. 8. Abundance and seasonal variation of the litter dwelling lizard *Morethia obscura* before and after a moderate-intensity autumn fire in jarrah forest.

no published data which address this issue (Friend 1993). Summer fires following prolonged drought could burn the habitat of some frog species such as the sunset frog (Roberts et al. *in press*).

Reptiles, especially lizards, are more resilient to fire than mammals because of their physiological adaptations to arid conditions. Most reptiles are burrowers with generalized invertebrate diets, and strongly seasonal activity and breeding patterns. Survival may be relatively high during a fire, but predation and starvation may be significant in the early postfire period (Newsome et al. 1975, Friend 1993). Species which inhabit jarrah forest leaf litter and which feed on invertebrates in the leaf litter (e.g., *Morethia obscura*) were affected for up to 2 years by high-intensity summer/autumn fires and recovered slowly as the leaf litter reestablished (Figure 8). Again, this species is likely to be less affected by patchy, low-intensity fires.

In southwest Australian ecosystems, invertebrates typically decline immediately after fire, but many species/groups return to prefire abundance and species diversity levels within 1–2 years. The level of taxonomic resolution (i.e., Orders versus species levels of identification) influences the interpretation of results of fire impact studies because individual species may be markedly affected (increase or decrease), but at the Order level of identification these trends will tend to cancel out and thus not be apparent. Studies of the responses of ground dwelling spiders in a Wheatbelt reserve (Strehlow 1993) showed that postfire succession does not necessarily return to the prefire state, but instead to a new transient state also occupied by undisturbed communities. This presents a problem for managers if the management aim is a fixed target in terms of biodiversity. Management goals need to reflect the reality, and aim to ensure no loss of biodiversity over a defined unit of space and time.

Invertebrate population responses are highly variable in space and time and seasonal factors such as rainfall often outweigh any effects due to fire (Friend and Williams 1996). Some invertebrate groups are more sensitive to fire than others. For example, long-lived and sedentary animals such as trap-door spiders (Myglamorphs) are vulnerable to fire and if extant animals are killed by fire, the species could become locally depleted for a period of time (Main 1987).

There appears to be a gradient in response to fire by invertebrates across major bioclimatic types (Friend 1995). Generally, fauna in the drier habitat types appear to be more resilient to single fire events than that in more mesic environments, perhaps reflecting the adaptations of the former for surviving in periodically arid environments. This suggests that sustained high fire frequencies (<6 year intervals) in wet southwest forests, particularly karri and tingle forests, could lead to localized reductions of some invertebrate groups, particularly if these fires are not patchy. This needs further study.

## CONCLUSIONS

Fauna and flora in southwest Australian ecosystems display a variety of physical and behavioral adaptations which enable them to persist in this fire-prone environment. However, based on the life histories of the most fire-sensitive taxa, they are not immune to any fire regime. Fire frequency and scale, or patchiness (which is linked to intensity and season), are the most critical factors to consider when planning fire regimes to conserve biodiversity. These factors are also vital for managing fuel loads and for reducing the threat of wildfires.

Information about potentially fire-sensitive taxa, characterized by plants with long juvenile periods and which depend on canopy stored seed for regeneration, and by animals which have very specific habitat requirements, such as the honey possum and various invertebrate groups (e.g., Mygalomorphs), can be used to set ecologically sustainable limits to fire frequency and scale. Based on these parameters, the sustainable minimum fire frequency for upland intermediate and high rainfall (<900 millimeters per year) jarrah forest is about 6 years, for low rainfall forest (750 millimeters per year) about 8 years, and for south coast mallee-heath (400 millimeters per year), about 15 years. These ecosystems will recover from occasional fires at shorter intervals, but we predict that sustained burning of the entire landscape (as opposed to patch burning) at intervals below these limits (i.e., Fire Frequency Ratio (FFR) <2) will eventually lead to the decline or loss of fire-sensitive taxa and increases in disturbance opportunists such as herbs and grasses.

The responses to fire by species such as the red-tailed phascogale and the honey possum provide clear evidence that fire scale, patchiness, and intensity are important determinants of postfire responses and the species' capacities to recover. While definitive data on the most appropriate scale at which to burn these ecosystems is unavailable, we suggest that large scale (>3,000 hectares), intense (> 2,000 kilowatts per year) fires at high to medium frequency (FFR 2–4) and which burn the entire landscape are least desirable, and if sustained, are likely to have severe local impacts on fauna particularly in the medium term. Remnant vegetation and extant or island populations and communities are particularly vulnerable to such a regime. Fuel reduction burns, where possible, should be restricted to blocks less than about 5,000 hectares, but more importantly, the burns should be patchy and low intensity. This is best achieved under early spring conditions when strong moisture gradients exist across the landscape. These conditions result in low-intensity fires and parts of the landscape being flammable (ridge tops, areas of high fuel loading) and other parts being non-flammable (areas with discontinuous surface fuel, riparian zones, etc.). Fire managers should aim for a mosaic of burned (60–70%) and unburned (30–40%) patches within these blocks. Special habitats such as riparian zones, granite outcrops, and other treeless, moisture gaining sites throughout the forests should be burned less frequently and therefore should not be burned during every fuel reduction burn, if at all possible. This can be achieved if prescribed burns are well planned and executed early in spring and when the Soil Dryness Index (Burrows 1985) is relatively low. If this cannot be achieved, and the terrain or weather conditions are such that the entire landscape will burn out, then blocks should be less than about 2,000 hectares. If fire managers attempt to retain very large, contiguous blocks of vegetation in an unburned state, then there is the real risk that these patches will burn completely and intensely under summer wildfire conditions with little chance of control. This will have a significant negative impact on wildlife, particularly fire-sensitive fauna, which may take many years to recover.

While low-intensity, patchy spring fires are recommended for fuel reduction and for maintaining biodiversity, we also recommend varying the season and interval between prescribed fires. Fires under relatively dry soil and fuel conditions in summer and autumn result in more complete consumption of live and dead vegetation and superior seedling regeneration and establishment. However, such fires should only be implemented occasionally (FFR >8), for the reasons discussed above, or if conditions dictate more frequent implementation (FFR <8), then the size of burned patches should be reduced.

The postfire response of flora and most small vertebrate groups are closely tied to and may be predicted from their regeneration strategy and juvenile period (flora) and shelter, food, and breeding requirements (faunal life history parameters, Friend 1993). Flora and fauna can therefore be grouped in terms of life form categories based on these life history parameters which, together with fuel accumulation and fire behavior knowledge, can be used to devise appropriate fire regimes in south-west Australian ecosystems to meet wildfire protection and conservation objectives. Based on this information, we suggest the fire regime shown in Figure 9 be applied to high rainfall jarrah forests (>900 millimeters per annum) to achieve conservation and wildfire protection benefits. The same regime could apply to other southwest ecosystems, but fire frequencies would need to be adjusted according to the biological indicators. Clearly, there will be localities where such a regime is not appropriate for various reasons. The Wildfire Threat Analysis (Muller

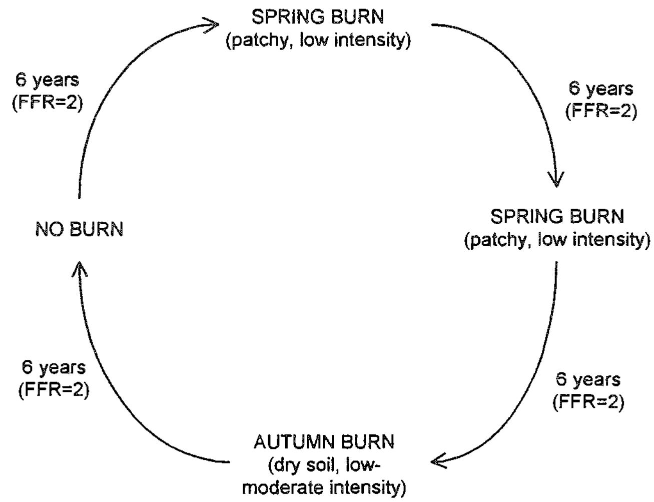


Fig. 9. A fire regime for intermediate and high rainfall jarrah forest based on biological indicators, and which incorporates seasonal and temporal variation to achieve conservation and wildfire protection benefits. FFR = fire frequency ratio.

1993), a systematic procedure used by the Department of Conservation and Land Management for analyzing wildfire threat and for determining mitigation strategies, would provide a sound basis for making these decisions.

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