DEPARTMENT OF
SUSTAINABILITY AND ENVIRONMENT

Ecological Burning in Box-Ironbark Forests: Phase 1 - Literature Review

Report to North Central Catchment Management Authority

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Arthur Rylah Institute for
Environmental Research

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Phase 1 – Literature Review

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EXECUTIVE SUMMARY

Box-Ironbark ecosystems comprise approximately 250 000 hectares of predominantly dry sclerophyll forest inland of the Great Dividing Range in Victoria. They are home to a diverse range of flora and fauna species, with 1330 vascular plant species, 213 bird species, 54 mammal species, 43 reptile species and 14 frog species recorded across the broader ecosystem.

Since European settlement, these forests have been extensively cleared for agriculture, urban development, timber cutting and mining. Much of the extant vegetation now exists as remnants of coppice regrowth with depleted understorey in areas of poorer soil, and this has had serious implications for ecosystem biodiversity.

The North Central Catchment Management Authority, as part of its Regional Catchment Strategy and Native Vegetation Plan, identified the need to “Develop and implement appropriate fire management regimes to sustain ecological processes in key private land vegetation remnants….” Given the lack of knowledge on such fire regimes, a review of current fire knowledge and practices was required, ultimately facilitating the creation of broad criteria for the establishment (or otherwise) of an appropriate ecological burning regime.

This report comprises Phase 1 of the Ecological Burn Project, the Literature Review, and focuses on the impacts of ecological burning in Box-Ironbark or similar ecosystems.

1. The review has not been able to shed light on historical aboriginal burning regimes. Some have suggested that seasonal burning was probably undertaken, but little direct evidence exists, and what evidence there is relates to landscape-wide observations that are not site- or vegetation-specific. Fire has probably played a minor part in influencing the vegetation structure and faunal assemblages in Box-Ironbark ecosystems. Management should now be geared towards the needs of the forests as they exist today, not as they existed in some idealised pre-European state.
2. The response of the understorey to applied burning will depend heavily on the nature of individual remnants, season, landscape position, soil type, seed bank, disturbance history and susceptibility to edge effects. Some species, particularly leguminous shrubs and short-lived obligate seeders, will be promoted by fire in the short-term. Resprouting species that make up a large proportion of the flora will be little affected unless burning is frequent. Few species rely on fire for germination, and most species that are stimulated by fire will still recruit at a low level in the absence of fire. No species should be lost through burning provided the inter-fire period allows all species to reach reproductive maturity (a minimum of 10-20 years) but absent species are highly unlikely to reappear. Most species will persist even when the interval between fires exceeds 50 years. Further research is required into the germination requirements of Box-Ironbark shrub species, and the effects of applied burning (taking advantage of DSE's fuel-reduction program).

3. The response of the canopy will depend on the intensity of the fire. In most instances, fire in small remnants is unlikely to be of sufficient intensity to lead to canopy replacement. In any event, most Box-Ironbark species show continual recruitment in the absence of fire, and thus do not rely on it. In any one patch, the minimum inter-fire period for a fire that kills or severely reduces the overstorey and that allows full recovery of structure is likely to be around 60 years. Further research is required to determine the germination requirements for Ironbark eucalypts.
4. Litter plays an important role in nutrient cycling, and provides important habitat for invertebrates and small vertebrates. Frequent burning (for example, 3-5 year intervals) will disrupt natural processes in the short-term and may eventually lead to a depletion of soil nutrients and loss of habitat.
5. Research into the effects of fire on invertebrates is confounded by high natural variability, and it is often difficult to determine the baseline or climax community. Short-term effects are intimately linked in many cases to burning of the litter layer, and burning at frequencies as high as every three years should be avoided. More research is required into the effects of fire on termites, known to be key drivers of secondary productivity in these forests.
6. The effects of fire on birds are strongly dependent on the severity of the fire and the structural components of the forest that are burnt, and recovery of populations is linked to the recovery of the vegetation. Some seed-eaters may be advantaged in the long-term if fire promotes new vegetation growth. Other species may be advantaged in the short-term by the availability of post-fire carrion or the reduction in protective cover for prey. However, species that rely on the ground layers for nesting or foraging may be disadvantaged in the short-term by low-intensity fire, particularly if it interferes with breeding. The minimum inter-fire period is likely to be similar to that which will allow full recovery of understorey structure (i.e. at least 25 years).
7. The effects of fire on mammals are also linked to the effects and recovery of the vegetation and the intensity of the burn. Some common herbivores take advantage of the flush of new plant growth, but small animals dependent on the ground layer will be disadvantaged in the short-term. Arboreal mammals are likely to be disadvantaged only if the fire is of high intensity. In general, it is believed that small mammal populations will not be disadvantaged, provided the minimum inter-fire period is at least 15-20 years. However, in isolated private remnants, recolonisation by small mammals after fire may be difficult. Further research is required, particularly in regard to bats and arboreal mammals such as possums.
8. Many reptiles and frogs are likely to be disadvantaged in the short-term by any burning due to their dependence on the litter and ground layers. Spring burning will affect the breeding of many reptile species, while autumn burning may affect the breeding of some frog species. In remnants, isolation is likely to be a barrier to recolonisation after fire. Further research is required on this fauna group.

INTRODUCTION

Box-Ironbark ecosystems comprise approximately 250 000 hectares of predominantly dry sclerophyll eucalypt forest on hills and slopes inland of the Great Dividing Range in northern Victoria (Calder *et al.* 1994; Kellas 1991; Woodgate and Black 1988) (Figure 1). Most of these forests and woodlands are located between the Divide and the Victorian Riverina, but small disjunct pockets also exist in rain shadow areas within lower-elevation mixed species forests, such as near Heyfield (eastern Victoria) and Airey's Inlet, south of Geelong (Kellas 1991). Box-Ironbark forests are generally found at elevations from around 200 to 300 metres above sea level, on slopes that (with the exception of some granitic areas) rarely exceed 10° (Newman 1961). Ordovician and Silurian sedimentary rocks are the most widespread in the area, and soils tend to be skeletal with poor water-holding capacity, and prone to gully erosion when ground cover is removed (Newman 1961). Average annual rainfall predominantly ranges from 400 to 700 mm, and temperatures are low in winter and high in summer (ECC 1997), often exceeding 35° C between October and March.

The area commonly referred to as 'Box-Ironbark' encompasses numerous floristic communities, including 11 eucalypt forest communities, 10 eucalypt woodland, 3 mallee and 1 rock outcrop community (Muir *et al.* 1995). These twenty-five floristic communities and a number of sub-communities occur within seventeen broader ecological vegetation classes (note that ECC (2001) recognised 73 Box-Ironbark EVCs totalling 496 000 hectares, of which 50 were mosaics or complexes for mapping purposes). Pure or mixed stands of Box [typically Grey Box (*Eucalyptus microcarpa*), Red Box (*Eucalyptus polyanthemos*) and Long-leaf Box (*Eucalyptus goniocalyx*)] and Ironbark [Red Ironbark (*Eucalyptus tricarpa*) or Mugga Ironbark (*Eucalyptus sideroxylon*)] tend to comprise the greatest proportion of the tallest stratum (Muir *et al.* 1995). Other canopy species may include Red Stringybark (*Eucalyptus macrorhyncha*, notably on ridges), Yellow Gum (*Eucalyptus leucoxylon*), Yellow Box (*Eucalyptus melliodora,* often on alluvial soils), White Box (*Eucalyptus albens*) or River Red Gum (*Eucalyptus camaldulensis,* around water courses).

Understorey vegetation may be shrubby, herb-rich, or grassy (Muir *et al.* 1995). Leguminous or epacridaceous species tend to dominate, with common shrubs including *Acacia*, *Pultenaea*, *Dillwynia*, *Daviesia*, *Grevillea*, *Brachyloma*, *Astroloma* and *Cassinia* (Newman 1961). The density and variety of these understorey species tend to increase as the soils get poorer, and undergrowth is most developed in the Red Ironbark/Red Box/Red Stringybark association on steeper slopes (Newman 1961). Where a grassy layer exists it is generally dominated by Silvertop Wallaby-grass (*Joycea pallida*) and Grey Tussock-grass (*Poa sieberiana*). A total of 1330 vascular plant species, including 322 weed species and 70 rare or threatened species, have been identified in the broader Box-Ironbark region (Muir *et al.* 1995).

Distributions of vegetation communities may be partly determined by soil (type or depth) and moisture availability, which itself is dependent upon aspect and topography. Productivity in these forests is low (Kellas 1991), but subtle changes in topography in Box-Ironbark forests may lead to dramatic changes in vegetation communities over short distances, sometimes involving a complete change in canopy species and understorey structure (Bennett *et al.* 1999). On steeper slopes and ridges, Red Ironbark, Red Box and Red Stringybark form stands to around 15 metres in height, but on better drained lower slopes and alluvial plains Grey Box and Yellow Gum begin to dominate in stands to around 24 metres in height (Kellas 1991). Contemporary distribution and composition of vegetation in the region may have been directed by land-use practices of local Aborigines (eg. burning, game hunting and food plant gathering over the last *circa* 40 000 years) (Sherwin 1996), but probably more so by widespread, intensive land-use by Europeans over the last 160 years (ECC 2000).

The Box-Ironbark forests are home to a range of fauna species. Some of these are widespread species, such as the Echidna (*Tachyglossus aculeatus*), Willie Wagtail (*Rhipidura leucophrys*) and Australian Magpie (*Gymnorhina tibicen*) (ECC 1997). Some are Eyrean (inland) species at the outer limit of their range [for example, Budgerigar (*Melopsittacus undulatus*), Rainbow Bee-eater (*Merops ornatus*), Gould's Sand Goanna (*Varanus gouldii*)], or Bassian (coastal or mountain) species at the inland limit of their range [for example, Sugar Glider (*Petaurus breviceps*), Crimson Rosella (*Platycercus elegans*), Powerful Owl (*Ninox strenua*)]. This co-occurrence of Bassian and Eyrean species has resulted in a particularly diverse fauna, with 213 bird species and 54 mammal species recorded in the Box-Ironbark study area (ECC 1997, 2001). There have also been 43 reptile species and 14 frog species recorded (ECC 1997, 2001).

Some species are considered to be 'stronghold' species where Box-Ironbark forests provide the most important part of their range (ECC 1997). These include the Squirrel Glider (*Petaurus norfolcensis*), Barking Owl (*Ninox connivens*), White-bellied Cuckoo-shrike (*Coracina papuensis*), Painted Honeyeater (*Grantiella picta*), Black-chinned Honeyeater (*Melithreptus gularis*), Diamond Fire-tail (*Stagonopleura guttata*), Grey-crowned Babbler (*Pomatostomus temporalis*), Fuscous Honeyeater (*Lichenostomus fuscus*), Brush-tailed Phascogale (*Phascogale tapoatafa*), Bush Stone-curlew (*Burhinus grallarius*), Swift Parrot (*Lathamus discolor*), Turquoise Parrot (*Neophema pulchella*) and Regent Honeyeater (*Xanthomyza phrygia*) (ECC 1997). The last five species mentioned were widespread at the time of European settlement but have since become 'Box-Ironbark stronghold species' as a consequence of a decline in their range (ECC 1997).

Rapid shifts in vegetation over short distances are reflected in the distribution of fauna species, with notable differences in wildlife communities between gullies and ridges (Bennett 1999; ECC 2001). Gullies support around one third more bird species than do slopes and ridges, and twice the density (Mac Nally *et al.* 2000b). Possums are also more widespread and abundant in gullies, indicating that moister drainage lines with greater availability of large trees are important areas for wildlife populations (Bennett 1999; Soderquist and Mac Nally 2000). Species composition will also change dramatically over time, with seasonal movement of tens of thousands of birds affected by forest flowering patterns (Bennett 1999). Private remnants, the subject of this literature review, thus play a vital role in the movement and conservation of these animals.

Pre-European community structure and composition of Box-Ironbark forests has been speculated upon but rarely quantified. Based on limited evidence, such as observations by early explorers, pre-European vegetation was likely to have been typically composed of widely spaced, large eucalypts with a highly diverse grassy to shrubby understorey (Calder *et al.* 1994). Tree density was believed to have been no higher than around 30 large (760 mm to 915 mm diameter) trees per hectare, including trees of up to 1520 mm diameter (Newman 1961). Abundant hollows and large logs would have provided habitat for numerous bird, reptile and mammal species. Indeed, some 25% of such species require the presence of hollows for roosting and nesting (ECC 1997).

Anthropogenic disturbances since the 1840s, particularly mining, timber harvesting and land clearing for agriculture, have contributed to a severe reduction in the extent and quality of Box-Ironbark forests (ECC 2001). Extant blocks of forest are often confined to areas with poorer soils that were less suitable for farming (Newman 1961), and even then most remnants occur as low forests of coppice regrowth (Kellas 1991), lacking topsoil, and usually with only a sparse herbaceous layer (Muir *et al.* 1995). In the Warby Range State Park, only 4% of total trees currently have a diameter exceeding 60 cm, while nearly 80% of trees are smaller than 40 cm (Adkins *et al.* 2005). Only three out of 159 trees recorded in that study have hollows, with clear implications for fauna habitat (Adkins *et al.* 2005).

Several flora and fauna species which are considered to have once occurred in these forests are now extinct in Victoria (ECC 2001; Muir *et al.* 1995; Silveira *et al.* 1997), including Spike Grass (*Elytrophorus spicatus*), Purple Eyebright (*Euphrasia collina* ssp. *speciosa*), the Eastern Quoll (*Dasyurus viverrinus*), Rufous Bettong (*Aepyprymnus rufescens*) and White-footed Rabbit-rat (*Conilurus albipes*).

There is, fortunately, a growing awareness by landholders in the Box-Ironbark region of the importance of the conservation values of these forests. A survey in 1996 found that over 60% of respondents, particularly those with smaller properties, believed that Box-Ironbark remnants were very important for habitat or wildlife corridors (Dettmann *et al.* 2000). Nonetheless, owners of larger properties ascribed greater importance to more utilitarian values, such as shade for stock, water table regulation, river protection and erosion control (Dettmann *et al.* 2000). Most landholders were also greatly concerned about potential management problems associated with Box-Ironbark remnants, as a haven for pest animals, a source of weeds, and as a fire hazard (Dettmann *et al.* 2000).

Box-Ironbark forests are considered not prone to recurrent fires, making them possibly atypical of dry, sclerophyllous vegetation in Australia (Calder *et al.* 1994). The extent of previous indigenous burning is largely unknown (ECC 1997; Meredith 1987), and the effects of fire on flora and fauna communities or on individual plant or animal species are also largely unknown. Indeed, the influence of fire on site productivity and biotic composition is often poorly understood in native vegetation, and the use of fire as a management tool is sometimes controversial (Williams *et al.* 1994). Nonetheless, based on evidence from similar vegetation types, it has been commonly assumed that ecological burning (as opposed to fuel-reduction burning) might be a useful tool for improving the condition of degraded native vegetation remnants and enhancing species biodiversity.

Accordingly, the North Central CMA, as part of its Regional Catchment Strategy and

Native Vegetation Plan, has identified the need to "*Develop and implement appropriate fire management regimes to sustain ecological processes in key private land vegetation remnants*…." Given the lack of knowledge on such fire regimes, a review of current fire knowledge and practices was required, ultimately facilitating the creation of broad criteria for the establishment (or otherwise) of an appropriate ecological burning regime.

The aims of the investigation were therefore to:

- Review available scientific literature
- Liaise as necessary with universities, government departments or other groups involved in relevant research or with historical information, including anecdotal or "gut feel"
- Source information from similar ecosystems in south-east Australia
- Use plant Vital Attributes to determine burn frequencies and gauge possible plant community responses
- Establish justification (or otherwise) for burning degraded remnants, including possible responses and risks
- Gather any available information on the likely direct and indirect impacts of ecological burning on vegetation structure and fauna habitat
- Identify gaps in knowledge
- Propose next steps to enable the North Central CMA to increase its knowledge of ecological burning

This report comprises Phase 1 of the Ecological Burn Project: the Literature Review.

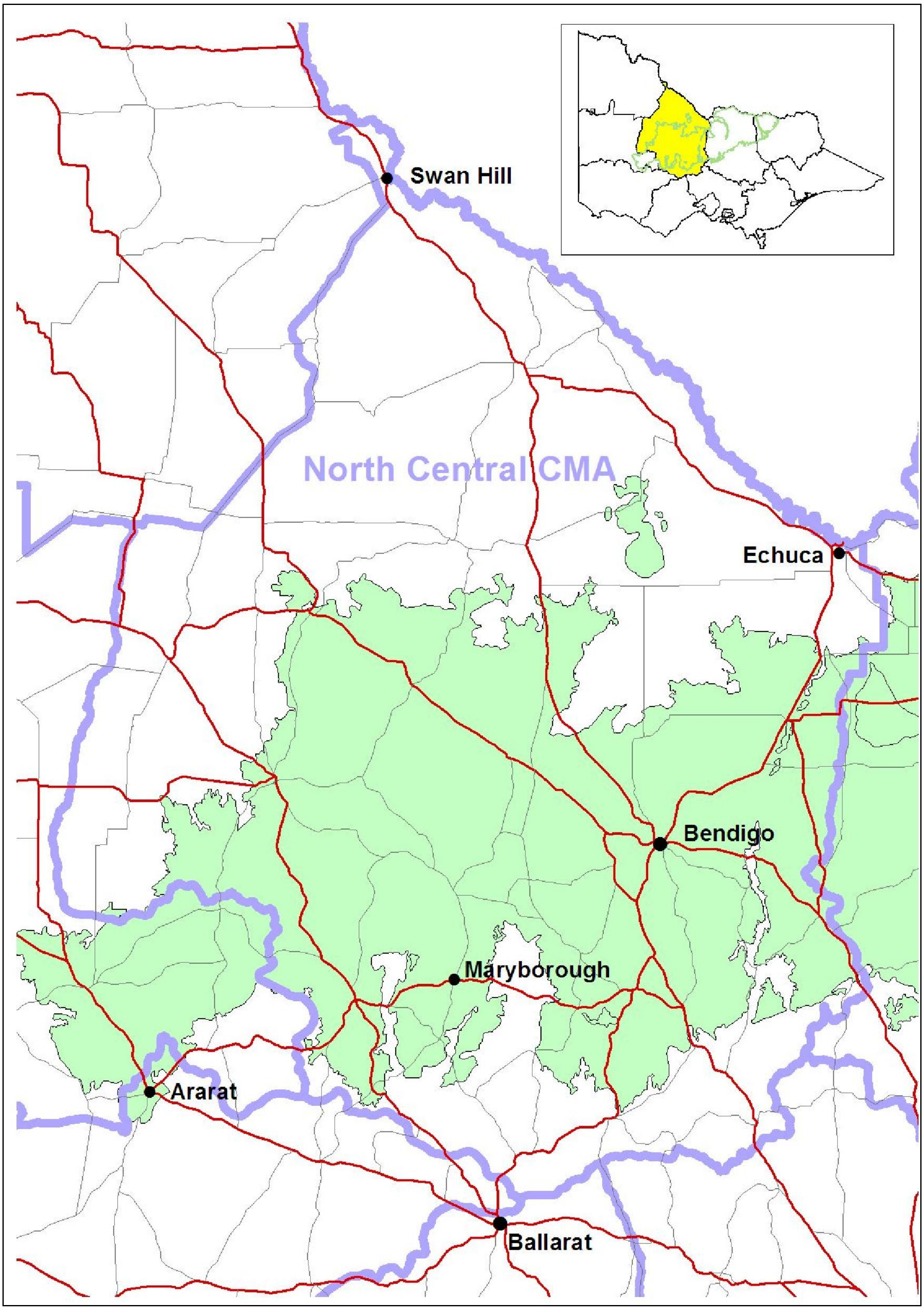

Figure 1. Map of North Central CMA, Victoria, showing the broader Box-Ironbark study area (as per ECC 2001). Note that forest cover is substantially less than that indicated on the map.

FIRE IN AUSTRALIAN ECOSYSTEMS

Role of Fire in Australia

For millennia, fire has been a natural feature of many Australian environments, a major driver of species composition and structure, and an important tool for ecosystem management (Bowman 1998; Chesterfield 1984; Kemp 1981; Meredith 1987; NRE 1999; Tolhurst 1999a). There is strong evidence from many Australian ecosystems for regular, deliberate burning since the arrival of Aboriginal colonists at least 40 000 years ago (Bowman 1998; Brown *et al.* 1998; Meredith 1987; Nicholson 1981; Singh *et al.* 1981), yet the frequency, extent and vegetation types involved remain largely unknown. Some authors have claimed that this burning was so extensive that it was a major driver of evolutionary diversification of the Australian biota, while others argue that it merely altered and maintained the geographic range of existing biota (Bowman 1998). The debate about indigenous burning and its relevance to present-day management has often been complicated by differences in the consideration of time scales, confusion about specific vegetation classifications, or management philosophies (h-skis 2000). Broad uncertainty in regard to the extent of indigenous burning across parts of Australia has led some authors to warn strongly against blindly accepting Aboriginal burning as a foregone certainty in every ecosystem, because evidence is often not there (Horton 2002).

Fires (ignoring those that are maliciously lit) can be divided into two broad groups, planned and unplanned, to distinguish those that are ignited intentionally by responsible people and those that are not (Meredith 1987). Unplanned fires (often caused by lightning strikes) are often called wildfires, while planned fires might include management burns (for various purposes), back burns or fuel reduction burns. Differences may exist in terms of fire intensity or season of burn, but in many cases the distinction between planned and unplanned fires is one of intention, not necessarily of fire behaviour or effect (Meredith 1987).

Fuel reduction burning aims to reduce the intensity of any future fire by reducing the amount of available fuel (Cheney 1996; DCNR 1995; Tolhurst and Cheney 1999), and is an important fire management tool in forests adjacent to communities or resources where protection is paramount. Fuel reduction burning may be compatible with ecological burning objectives (NRE 1999), but there may also be trade-offs (Meredith 1987). In tracts of forest classified as Management Zone 1 there is an acknowledgement that intensive burning to provide the highest level of strategic protection will have ecological impacts (DCNR 1995; DSE 2003b). Paradoxically, frequent fire may sometimes be counter-productive, in that it may change a forest understorey from predominantly herbaceous to predominantly shrubby (Chesterfield 1984; Tolhurst 2003; Wakefield 1970). Thus, repeated prescribed burning in some vegetation types may well be maintaining dense, lower diversity, more flammable understoreys (Chesterfield 1984; Tolhurst 2003).

There is growing awareness of the role of fire for ecological purposes, under the assumption that such ecological burning might be a useful tool for improving the condition of degraded native vegetation remnants and enhancing species biodiversity

(Friend *et al.* 1999; NRE 1999). However, this role remains controversial, as the precise influence of fire on biotic composition is often poorly understood (Friend 1993; Good 1981; Williams *et al.* 1994). While some managers are increasingly focused on promoting heterogeneity in burning patterns ("pyrodiversity begets biodiversity"), desired fire mosaics remain unspecified and the ecological significance of different burning patterns remains unknown (Parr and Andersen 2006). This has led to a situation where some fire-management plans may be based on pyrodiversity rhetoric that lacks substance in terms of biodiversity outcomes (Parr and Andersen 2006).

Therefore, a clear distinction must be made between fuel reduction burning (with the simple aim of reducing the fuel available for consumption in a future fire) and ecological burning (with the more esoteric aim of enhancing biodiversity). This report will focus on the latter.

Plant Adaptations and Responses to Fire - a General Overview

Plants differ widely in their ability to tolerate fire (Gill 1981; NRE 1999). Some are considered fire-sensitive, in that they are killed by a single fire. These plants have no mechanism to regenerate *in situ*, and must rely on the availability of propagules in adjoining unburnt areas. Fortunately, few Victorian plants are in this category, being found mostly in areas where fire is very infrequent, such as cool temperate and warm temperate rainforests, and to a lesser extent alpine communities.

Most plants in fire-prone regions such as the 'Mediterranean' ecosystems have evolved various strategies to cope with inevitable fire, with patterns of response that may be attributed to a large part to a combination of regeneration niche, disturbance frequency or resource gradients (Clarke and Knox 2002; Clarke *et al.* 2005; Pausas and Bradstock 2007). Two broad strategies for persistence are vegetative regeneration (resprouting) and seed regeneration (Gill 1981). Plants that resprout following fire have buds on the adult plants that survive that fire, and may be considered fire-resistant. These buds may be subterranean (e.g. on roots, rhizomes, lignotubers) or aerial (e.g. epicormic or deep in tussock bases), and they form new shoots when the top of the plant is killed and associated inhibitory factors are removed (Gill 1981). This group of plants is considered to be the least sensitive to loss through fire (Meredith 1987), although vitality and bud survival may nonetheless be reduced after fires at too high a frequency or intensity (Gill 1981).

Many plants are killed by fire, and regenerate by seed. These seeds may be stored for various periods of time in the soil (e.g. *Acacia*, pea species) or on the parent plant (e.g. *Eucalyptus*, *Banksia*, many *Hakea*) (Gill 1981), or be dispersed in from nearby areas. In the absence of fire, germination of the seeds of fire-responsive species is often uncommon, as the seeds require stimulation in the form of direct heat, chemicals in smoke or ash, or changes in light regime (Bell 1999). However, after fire there may be a strong response (Bell 1999; Brown *et al.* 2003; Gill 1981). A third category of regeneration strategy involves a combination of both resprouting and reseeding, depending on local fire severity.

Seed regenerators in particular are sensitive to fires at inappropriate intervals, and may be lost from an area if they have been unable to reach reproductive maturity before a subsequent fire (Gill 1981; Noble and Slatyer 1981). In contrast, where fire frequency is too low, some species may senesce, and their seed stores will eventually disappear (Noble and Slatyer 1981). These may also be lost from the community. For example, over the 200 years since settlement, a patch of coastal vegetation near Ocean Grove in southern Victoria has changed from an open grassy woodland, to an open scrub of *Acacia pycnantha* (Golden Wattle), to a closed scrub of *Allocasuarina littoralis* (Black Sheoak) (Lunt 1998). These changes were attributed to a long-term absence of fire. In general, however, plants are more likely to be lost through too frequent, rather than too infrequent, fire (Meredith 1987).

The importance of regeneration strategy (persistence after fire), combined with factors such as the longevity of each of the life-stages and tolerance of competition and site conditions, has led to the development of the Vital Attributes scheme (Noble and Slatyer 1981; Tolhurst 1999b). Three groups of vital attributes are recognised (Noble and Slatyer 1981):

- The method of species arrival after, or persistence during, a disturbance. This takes into consideration whether the plant is a reseeder or resprouter, or capable of both modes. It also considers whether seeds are stored in the soil, on the plant or off-site, or whether buds are aerial or subterranean.
- The ability to establish and grow to maturity in the developing community (tolerance). This identifies the conditions required by each species for establishment, such as elimination of competition. These first two attributes combine to give a 'species type'. While 30 such combinations are mathematically possible, in reality there are only 14 distinct patterns.
- The time taken for the species to reach critical life stages. Of particular importance are the time to reproductive maturity, life span, and longevity of the soil seed store. Local extinction occurs after plants have died and the seed store has decayed.

Relevant data are being accumulated by various researchers (Friend *et al.* 1999), and incorporated into a comprehensive database, to help model and predict the response of individual plants and vegetation communities to fires of various frequency. These data can be used to identify key fire response species, which in turn leads to an understanding of tolerable fire frequencies for the various vegetation types (Tolhurst 1999b; Tolhurst and Friend 2003). They also allow the modelling of the idealised age distribution for that vegetation (Friend *et al.* 1999).

The response of many reseeding plants is well known, being based on relatively simple metrics relating to parameters such as the time to reach reproductive maturity. Far less is known in regard to resprouting plants, and the fire frequency that would cause a potentially fatal reduction in plant vigour or bud viability.

Regardless of regeneration strategies, many extrinsic factors also have an impact on plant survival, such as the season of burn, intensity of burn, rainfall (or lack of) after the fire, frost, post-fire herbivory and seed-harvesting by animals. The effect on plants of the season of burn is strongly related to their Vital Attributes. For example, a fire at the time of seed-set would be detrimental to reseeding species. Conversely, a late-autumn burn might prevent resprouters from supplementing their stored carbohydrate (energy) levels for the oncoming winter (Tolhurst 2003; Tolsma 2002). Grazing animals are often drawn to burnt areas due to the availability of highly-palatable new growth, and post-fire grazing can have an adverse effect on subsequent regrowth, particularly if the burn area is small and grazers are not ‘saturated’ (Leigh and Holgate 1979; Meredith 1987). Kangaroos have a preference for new growth after fire, and a fire/grazing combination might cause a greater change in floristic composition or structure than fire or grazing alone (Robertson 1985). New shoots of *Poa* tussock-grass are preferentially grazed following burning in dry sclerophyll forest, as are new shoots of shrubs (Tolhurst and Oswin 1992). This is more pronounced following autumn burning, probably as a consequence of other green fodder being limited.

Fire intensity, if too low, may not stimulate adequate germination of seeds that require a particular degree of heating (Meredith 1987). In contrast, too high a fire intensity may damage the seeds or kill the necessary buds on resprouting plants. An appropriate amount of rainfall after the fire event is necessary for regeneration (Purdie 1977a), but heavy rain may erode the soil or wash away the propagules (Meredith 1987). All of these extrinsic factors may act to shape the vegetation community in the years following fire, by altering the relative abundances of individual species (Meredith 1987).

While fire is generally treated as the most important agent for mass-recruitment events in similar ecosystems in Australia (Auld and Tozer 1995; Benwell 1998; Roche *et al.* 1998), inter-fire recruitment (in the absence of fire) may be more common than previously thought (Purdie 1977b). For example, recruitment of *Eucalyptus* species after fire is generally highly obvious and vigorous. However, these species may not *need* recurrent fire to persist (Gill 1981). For many species, favourable environmental conditions (eg. diurnal light regime with adequate rains), may by sufficient to cue their germination (Bell 1999). However, high levels of herbivore grazing and browsing, along with other, largely anthropogenically-derived modifications to vegetation, may restrict recruitment success during such climatic opportunities. For example some *Eucalyptus* species may germinate after heavy autumn rains, but there may be high mortality of seedlings where grazing and/or browsing animals occur (Kellas 1991). Hence, observations of inter-fire recruitment may be difficult even where it occurs.

Faunal Adaptations and Responses to Fire- a General Overview

The impacts of fire on fauna (vertebrate or invertebrate) may be both direct and indirect, and depend heavily on the scale and intensity of the burn. As with plants, effects may be seen at both species and community levels, and over short or long time scales. Further, effects measured can be 'real' and attributable to fire, or 'artificial' and attributable simply to a greater chance of detection, adding a degree of complexity to the analysis of post-fire fauna data.

The direct and immediate effects of fire on fauna depend partly on the ability of individual animals to avoid that fire, and partly on the ability of survivors to find suitable habitat in or near the burnt area. Mobile animals, such as birds, bats and macropods, may be able to escape from the advancing fire if the fire front is moving relatively slowly, or if the fire is not of very high intensity (Brown *et al.* 1998;

Suckling and Macfarlane 1984). Burrowing animals may be protected from the hottest fires by being well underground, while animals that are dependent on tree hollows may only be protected from 'cool' ground fires. However, there are few data on this (Meredith 1987). Less-mobile animals, including invertebrates, may survive only in unburnt patches or microsites, and suffer high mortality. In Australian ecosystems, most information is available for small mammals, which often have a predictable and consistent response to fire. There are fewer data sets for other fauna types.

The indirect effects of fire, through its impact on habitat and plant resources, will also have an acute effect (both spatially and temporally) on many fauna, depending on the severity, extent and frequency of fire (which are influenced by topography, fuel load and weather conditions). Effects will also differ substantially according to the resource needs of individual species, and the extent to which the post-fire environment can provide those needs. In most instances, mammal response to fire is determined more by vegetation density and structure than by time since fire *per se* (Friend and Wayne 2003).

Fuel reduction burns may affect mostly ground-dwelling and understorey birds (Christensen and Kimber 1975), while intense wildfire will drastically affect birds from all levels of the forest (Loyn 1997). Fire frequency will also impact indirectly on bird communities through its long-term impact on vegetation structure. For example, if fire is too frequent to allow a mature forest plant community to develop, then specialist birds requiring mature forest will be lost. Similarly, some mammal species may rely on mosaics of vegetation of particular age classes (Meredith 1987; Possingham and Gepp 1996). In general, most small mammal species in dry forests are unlikely to be disadvantaged by fires where the inter-fire period is at least 15-20 years (Meredith 1987).

FIRE IN BOX-IRONBARK ECOSYSTEMS

Historical Fire Regimes in Box-Ironbark Forests

The determination of historical fire regimes in Box-Ironbark forests prior to European settlement is fraught with difficulty. Some suggest that regular, seasonal indigenous burning was probably undertaken in the region as a support for hunting (Calder *et al.* 1994) and for maintaining the supply of vegetable food (mostly plant roots) (Gott 2005). Others suggest that these forests were not deliberately burnt. However, little direct evidence exists either way (ECC 1997, 2001), and what evidence there is relates to landscape-wide observations that cannot be referred to specific sites. We are thus left with incomplete and sometimes contradictory historical notes and anecdotal evidence.

Box-Ironbark forests, particularly near settlements, have clearly experienced more frequent fires since European settlement. For example, of 1849 wildfires across the Bendigo Fire District between 1983/84 and 2002/03, only 7.2% were known to have resulted from lightning strike: the remainder were deliberate, accidental or of unknown origin (DSE 2003b). Some of the increase in fire since settlement has been due to a purported need for prescribed burning for human protection from wildfire, to reduce the biomass of understorey vegetation that would usually carry a wildfire. However, the argument for such prescribed burning in Box-Ironbark forests has not been well supported in the scientific literature. With perceived low rates of accumulation of litter and flammable vegetation (Calder *et al.* 1994), fuel hazard levels in Box-Ironbark forests have been considered by some authors to be generally low, with the exception of areas containing Stringybark species (DSE 2003b; ECC 1997). Overall fuel hazard levels may still be high in some areas, depending on canopy species, shrub cover and the previous disturbance history of a site (Chatto 1996). Nonetheless, these forests may be atypical of dry sclerophyllous vegetation in Australia in that they are generally considered not prone to recurrent fire (Calder *et al.* 1994). This view is supported by historical evidence from White Box (*Eucalyptus albens*) woodlands in eastern Victoria, where fires caused by lightning strikes did not generally carry far, and no attempts were made by Aborigines to deliberately fire those woodlands (Wakefield 1970).

Given that Box-Ironbark forests are not considered fire-prone, fire has probably played a minor role in shaping the Box-Ironbark ecosystem since European settlement (ECC 1997, 2001). Box-Ironbark forests now exist as remnants within an agricultural landscape. Some of these remnants have rarely burned, in part because their isolation has reduced the likelihood of fire spreading from surrounding areas. Fires that have occurred have usually been small in size and low in intensity compared to those that have occurred in other forest types. For example, between 1980/81 and 1995/96, almost 80% of the 800 wildfires recorded on public land across the Box-Ironbark region were less than 5 ha in size, and only around 15 exceeded 100 ha (ECC 1997, 2001).

Nonetheless, fire management is still considered by Parks Victoria to be an important strategy in the various Box-Ironbark parks. The aims of fire management include protecting human life, property and park values, protecting surrounding land from fires spreading from the park, maintaining fire regimes appropriate to the conservation of native flora and fauna, and minimising the adverse effects of fire and fire suppression methods (NRE 1996; PV 1997, 2005, 2006a, b, 2007). The Bendigo Fire District fire protection plan recognises the use of fire in Fuel Management Zone 4 (for specific flora and fauna management), but defers to individual management plans for formulation of appropriate fire regimes (DSE 2003b). However, the justification for ecological burning is not elaborated in current park management plans, nor is there any indication of an appropriate fire regime, reflecting the paucity of knowledge in this area. Thus, these management plans can only acknowledge the need for monitoring of the effects of burning on plant communities and wildlife habitat, and encourage studies into the requirement for ecological burning for vegetation management (NRE 1996; PV 1997, 2005, 2006b).

The ecological disturbance associated with a long period of European settlement makes retrospective deduction of indigenous burning regimes difficult. However, from a practical perspective, the issue of pre-European fire regimes may be seen as largely academic (Bowman 1998; Meredith 1987). Much of the distribution and composition of vegetation and forest structure, and faunal assemblages, have been influenced by widespread, intensive land-use by Europeans over the last 160 years (ECC 2000; Henderson and Keith 2002). It is thus impractical to attempt to reinstate historical burning or other management regimes that may have been applied under vastly different conditions (Bowman 1998; Horton 2002; Williams *et al.* 1994). Ecological fire management should therefore be based on the likely responses of extant flora and fauna assemblages, under conditions that exist at present. In the following sections we will attempt to address those likely responses.

Key findings

- The review has not been able to shed light on historical aboriginal burning regimes. Little direct evidence exists, and what evidence there is relates to landscape-wide observations that are not site- or vegetation-specific.
- Box-Ironbark forests are not considered fire-prone, and most fires that occur now tend to be small in size and probably low in intensity.
- Fire is likely to have played a minor part in influencing the vegetation structure and faunal assemblages in these forests.
- The justification for ecological burning is not elaborated in current management plans.
- The structure and diversity of Box-Ironbark forests have now been altered by a long history of post-Eropean disturbance.
- Management should be geared towards the needs of the forests as they exist today, not as they existed in some idealised pre-European state.

Effects of Fire on Understorey

Dry sclerophyll forest tends to be tolerant of fire, being able to recover rapidly even after intense fire (Christensen *et al.* 1981). Pyric succession is determined to a large extent by pre-fire composition, hence does not follow the classical successional changes documented for many northern hemisphere ecosystems (Christensen *et al.* 1981). A major proportion of the species are able to regenerate vegetatively, such that the basic distribution of species changes little after fire.

The short-term response of reseeding plants to fire is often intimately linked to seed germination requirements, such as direct heat, chemicals in smoke or ash, or changes in light regime (Bell 1999). There is a wealth of available data on the requirements for individual genera or species, including many commonly found in Box-Ironbark forests. For example, seeds of species of *Acacia*, *Daviesia* and *Pultenaea* are stimulated by direct heat shock, while species of *Hibbertia* and *Lomandra* are stimulated by smoke (Bell 1999). However, there are little solid data to indicate if fire is a necessary pre-requisite for germination, or if other cues might suffice. Five species of *Acacia* from Box-Ironbark were found to germinate in the absence of fire (Orscheg 2006). Only *Pultenaea prostrata* (Silky Bush-pea) in that study appeared to have some reliance on fire (or other disturbance), as all seed remained dormant after three years of burial (Orscheg 2006).

Only two studies were found that specifically looked at the effects of fire on the understorey of Box-Ironbark forests of the type found in the study area, that of Orscheg (2006) and a minor student report by Hartskeerl (1997). No study was found that addressed the short and long-term changes in Box-Ironbark plant communities as a result of repeated fire. Of most relevance is likely to be the comprehensive fire effects study undertaken in the Wombat State Forest, north-west of Melbourne. This is a mixed, eucalypt foothill forest, with canopy species including *Eucalyptus obliqua* (Messmate Stringybark), *E. rubida* (Candlebark) and *E. radiata* (Narrow-leaved Peppermint) (Tolhurst 1992). Understorey structure varies from that commonly found in Box-Ironbark forest, with *Pteridium esculentum* (Austral Bracken) and *Tetrarrhena juncea* (Forest Wire-grass) contributing over two-thirds of total understorey cover. However, shrub genera such as *Daviesia*, *Acacia* and *Pultenaea* and many herbaceous species are common to both forest types.

No plant species were lost or gained in the Wombat State Forest as a result of up to four successive spring fires or three autumn fires (Tolhurst 1996, 2003), although changes in the relative cover/abundance varied between species. Total understorey cover in the year after fire decreased with each consecutive spring burn, suggesting that three years was insufficient time for plants to recover fully (Tolhurst 2003). Autumn burning affected total understorey cover to a greater extent, and it had not fully recovered after ten years. Changes were also evident in understorey structure for at least four years post-fire because of the reduced height of woody plants (Tolhurst 2003).

The average cover-abundance of understorey plants over the the most recent five years of the Wombat State Forest experiment were compared to the average cover-abundance over the first five years of the experiment (1985-1989), to determine the response of individual species to burning treatment. As expected, individual species showed different responses depending on the combination of fire frequency and season (Table 1, Kevin Tolhurst, The University of Melbourne, unpublished data), and few generalisations were possible.

Table 1. Response of some understorey vascular plants to repeated burning in the Wombat State Forest, Victoria (unpublished data, Kevin Tolhurst, The University of Melbourne). Species are only shown when their abundance increased or decreased by at least one Braun-Blanquet cover class, and the list is therefore not exhaustive. Key: A3 = autumn burning at short rotation (≈ 3 years), A10 = autumn burning at long rotation (≈ 10 years), S3 = spring burning at short rotation, S10 = spring burning at long rotation. * denotes weed species.

Scientific name	Common name	Control	A3	A10	S3	S10
Acacia aculeatissima	Thin-leaf Wattle					--
Acacia dealbata	Silver Wattle				--	
Acacia melanoxylon	Blackwood	+		--		--
Acacia mucronata	Narrow-leaf Wattle	--				
Acacia stricta	Hop Wattle	+		+		--
Acaena novae-zelandiae	Bidgee-widgee				--	
Acrotriche prostrata	Trailing Ground-berry	+	--		+	+
Acrotriche serrulata	Honey-pots					--
*Anthoxanthum odoratum**	Sweet Vernal-grass	--	--		--	+
Anthropodium milleflorum	Pale Vanilla-lily	+	+			+
Asperula scoparia	Prickly Woodruff	+				
Austrodanthonia penicillata	Slender Wallaby-grass		--	+	--	--
Austrodanthonia pilosa	Velvet Wallaby-grass	--		+	--	--
Billardiera scandens	Common Apple-berry	--		+		--
Brunonia australis	Blue Pincushion					--
Burchardia umbellata	Milkmaids			--		+
Chiloglottis gunnii	Common Bird-orchid				+	
Clematis aristata	Mountain Clematis				--	
Daviesia leptophylla	Narrow-leaf Bitter-pea	+				
Desmodium varians	Slender Tick-trefoil				+	
Deyeuxia minor	Small Bent-grass		+			
Deyeuxia quadriseta	Reed Bent-grass	--				
Dichelachne micrantha	Small-seed Plume-grass	--		--	--	
Dillwynia cinerascens	Grey Parrot-pea		--			
Dipodium punctatum	Hyacinth Orchid		+			
Drosera peltata	Pale Sundew	+				+
Echinopogon ovatus	Common Hedgehog-grass				+	
Epacris impressa	Common Heath			--		
Eucalyptus obliqua	Messmate Stringybark	--		--	--	+
Eucalyptus radiata	Narrow-leaf Peppermint			--	--	
Eucalyptus rubida	Candlebark	+	--	--	--	
Euchiton collinus	Creeping Cudweed	--	+			--
Geranium potentilloides	Cinquefoil Cranesbill		--		--	
Glycine clandestina	Twining Glycine				+	
Gonocarpus tetragynus	Common Raspwort	--	--	--	--	+
Goodenia lanata	Trailing Goodenia	--	+	--	--	+
Hardenbergia violacea	Purple Coral-pea				+	--

Helichrysum scorpioides	Button Everlasting		--		+	
Hydrocotyle hirta	Hairy Pennywort	--	--	--	--	
Hypericum gramineum	Small St John's Wort	--	+		--	+
Hypochoeris radicata	Cat's Ear	--			--	
Indigofera australis	Austral Indigo				--	
Joycea pallida	Silvertop Wallaby-grass	+				
Lagenophora stipitata	Common Bottle-daisy			+	+	
Lomandra filiformis	Wattle Mat-rush	--				--
Lomandra longifolia	Spiny-headed Mat-rush		+			+
Luzula meridionalis	Common Woodrush	+				+
Microlaena stipoides	Weeping Grass			--	--	
Microseris scapigera	Yam Daisy	+				
Olearia erubescens	Moth Daisy-bush	--	+			
Opercularia varia	Variable Stinkweed	+	+			
Picris angustifolia	Native Picris				--	
Poa labillardierei	Common Tussock-grass				+	
Poa sieberiana	Grey Tussock-grass	--	+		--	--
Poranthera microphylla	Small Poranthera	--		+		+
Pteridium esculentum	Austral Bracken	--			--	
Pultenaea gunnii	Golden Bush-pea	--				
Pultenaea muelleri	Mueller's Bush-pea	--				
*Rubus fruticosus**	Blackberry				--	
Senecio minimus	Shrubby Fireweed		--	--	--	
Stellaria pungens	Prickly Starwort				--	
Stylidium graminifolium	Grass Triggerplant					--
Tetrarrhena juncea	Forest Wire-grass	--			--	--
Thysanotus tuberosus	Common Fringe-lily	+	+		+	+
Viola betonicifolia	Showy Violet				--	
Viola hederacea	Ivy-leaf Violet	--	--	--		--
Wahlenbergia stricta	Tall Bluebell	--	+		+	+

Most *Acacia* and *Eucalyptus* species declined after several fires (with the exception of *Acacia stricta* in the long-rotation autumn treatment and *Eucalyptus obliqua* in the long-rotation spring treatment), reflecting the time required to reach reproductive maturity or the age at which saplings can survive the fire. Some herbaceous species also declined when subjected to short-rotation autumn or spring burning, such as *Austrodanthonia penicillata, Geranium potentilloides, Gonocarpus tetragynus, Hydrocotyle hirta* and *Senecio minimus.* Several species were able to increase when subjected to short-rotation burning, but with the exceptions of *Thysanotus tuberosus* and *Wahlenbergia stricta* the increase was highly dependent on the season of burn. The data also show that occasional burning may be a useful tool for stimulating the abundance of various species, because 23 species declined in the long-unburnt controls compared to 13 species that increased. For example, *Austrodanthonia pilosa* and *Billardiera scandens* decreased in the control and long-rotation spring treatments, but increased in the long-rotation autumn treatment. *Wahlenbergia stricta* increased in three of the four burning treatments (remaining stable under long-rotation autumn burning), and only decreased in the unburnt control, suggesting it is clearly advantaged by occasional fire. Interestingly, *Microseris lanceolata* (Yam Daisy), a species traditionally encouraged by aboriginal burning, increased only in the unburnt control treatment, remaining stable under the four sequential burning treatments, perhaps as a result of post-fire rabbit grazing.

These data are invaluable in demonstrating the need for robust, long-term experimental data that are not drawn from a single fire event. *Acacia* species, for example, are stimulated by fire (Bell 1999; Brown *et al.* 2003; Heislers 1974; Orscheg 2006; PV 1997), but the increases observed after single fires are generally not seen after several sequential fires (Table 10).

In dry sclerophyll forest near Canberra [dominated by *E. macrorhyncha* (Red Stringybark), *E. rossii* (Inland Scribbly Gum) and *E. mannifera* (Brittle Gum)], the floristic composition was not changed by a single fire, and 12 months post-fire all species were represented by either resprouts or seedlings (Purdie 1977b), albeit with some initial (and inconsistent) differences in the dominance of tall shrub species. These differences were attributed to factors such as mode of regeneration, available seed reserves, germination requirements and susceptibility to mortality. Fire stimulated germination of most species, particularly leguminous species such as *Acacia genistifolia* (Spreading Wattle), *Dillwynia phylicoides* (Small-leaf Parrot-pea), *Pultenaea procumbens* (Heathy Bush-pea) and *Daviesia mimosoides* (Blunt-leaf Bitter-pea), although seedling mortality was high during the first year (Purdie 1977b).

More positive results were obtained in Cumberland Plain Woodland, a Box-Ironbark woodland dominated by *E. moluccana* (Grey or Gum-topped Box), *E. tereticornis* (Forest Red Gum) and *E. crebra* (Narrow-leaved Red Ironbark) around Sydney, NSW (Hill and French 2004). Eighteen months after a single burn in summer or late winter/early spring, the richness and abundance of shrubs was higher in burnt than unburnt treatments, leading the authors to suggest that fire was a useful management tool for promoting increased diversity.

Shrub recruitment (of existing species) appeared to be stimulated by a single fire in the Wombat State Forest, although cover then declined to below pre-fire levels, and a fire-free period of at least 10 years appears to be required if shrubs are to continue to expand their occupation of the site (Tolhurst 2003). Shrub cover, particularly of *Acacia aspera* (Rough Wattle) and *A. mearnsii* (Black Wattle), had increased dramatically three years after the 1969 wildfire in Box and Stringybark forest near Daylesford, from 10% to 27% on ridges, and from 5% to 34% in gullies (Heislers 1974).

Fire in 1985 stimulated the development of dense *Acacia pycnantha* (Golden Wattle) in the Paddys Ranges (Box-Ironbark) park near Maryborough (PV 1997). In the absence of further disturbance these shrubs are gradually thinning through natural processes, but a large soil seed bank will remain. Some areas in the same park contain dense thickets of senescent *Acacia paradoxa* (Hedge Wattle), presumably stimulated by the same fire (Arthur Rylah Institute, unpublished data). The response by *Acacia* is not surprising, as five species studied in Box-Ironbark forest near Bendigo [(*Acacia ausfeldii* (Ausfeld's Wattle), *A. williamsonii* (Whirrakee Wattle), *A. genistifolia*, *A. pycnantha* and *A. paradoxa*] show a strong heat-stimulated germination response (Brown *et al.* 2003). However, as already indicated, the response of shrubs to multiple fires may be substantially different to their response from a single fire. In dry forests in New South Wales, with overstorey dominated by *Angophora costata* (Smooth-barked Apple) and *Corymbia gummifera* (Red Bloodwood), species richness in the shrubby understorey was higher in fragments burned five and six years earlier than it was in fragments burned 10 years or more ago (Ross *et al.* 2002). Large increases in shrub cover were also noted in Peppermint forests in eastern Victoria that were historically burnt at around 3-4 year intervals (or "as often as possible"), with prolific germination of genera such as *Olearia*, *Helichrysum* and *Cassinia* (Wakefield 1970). Continual burning of *E. sieberi* (Silvertop Ash) forest in Gippsland also led to the development of dense stands of leguminous shrubs (Incoll 1940, cited in Chesterfield 1984). However, these findings do not contradict those from the Wombat State Forest research: they merely highlight the differential role of plant vital attributes, particularly time-to-reproductive-maturity, in the response of individual species.

Promotion of shrubs by fire (particularly those that are able to reach reproductive maturity between fires) appears to be more prevalent in drier forests, because in more productive open eucalypt forests, such as on Fraser Island (Queensland), frequent fire has tended to reduce midstorey cover (Spencer and Baxter 2006). Similarly, frequent burning in conjunction with grazing has led to a substantial simplification of the woody understorey in temperate eucalypt forest in northern NSW (Henderson and Keith 2002). A year after autumn fuel-reduction burning in *Eucalyptus obliqua* (Messmate Stringybark) forest near Gembrook, Victoria, 33% of the understorey species recorded prior to the fire had not yet reappeared (Hamilton *et al.* 1991), despite cover having restored to around 40% of the pre-fire amount. One of the absent shrub species, *Goodenia ovata* (Hop Goodenia), is relatively common in Box-Ironbark forest. However, several shrubs increased substantially in density as a result of the fire, including *Hakea sericea* (Bushy Needlewood), *Acacia genistifolia* and *Platylobium formosum* (Handsome Flat-pea) (Hamilton *et al.* 1991), species that are also common in Box-Ironbark.

Landscape position can be a key determinant of fire effects on ground cover. The cover of herbs and grasses on ridges was around 10% prior to the single wildfire near Daylesford, and had recovered to 8.5% three years later (Heislers 1974). In contrast, the cover in gullies was 20% prior to fire, but had increased to 65% three years later. Ground cover in productive, open eucalypt forest on Fraser Island was similarly promoted by frequent fire (Spencer and Baxter 2006). This suggests that more productive parts of the landscape with species mix adapted to higher moisture and nutrient availability will respond differently to areas with fewer resources.

Fire frequency and season are also key determinants of the response to fire by ground cover. For example, in the Wombat State Forest, the cover of herbs in burnt plots exceeded that in the unburnt control plots within one or two years of a spring burn, and within four years of an autumn burn (Tolhurst 1996). Without burning, herb cover and density declined (Tolhurst 1996), although this may have been due to factors other than the absence of burning. However, herb cover declined to a greater extent following a second spring burn at short rotation, took a year longer to recover than after a single burn, then did not exceed the cover in the unburnt plots (Tolhurst 1996). A single autumn burn stimulated increased tillering in *Poa*, although regenerative capacity in terms of cover and height was lower than after a spring burn (Tolhurst 1996). Overall, the cover of tussocks was dramatically affected by burning, and remained at only 40% of the cover in the unburnt plots one year after first burning (Tolhurst 1996). Tussock cover had increased to 80% of that in the unburnt treatment four years after a single autumn or spring burn, but recovery was substantially slower following a second spring burn at short rotation (Tolhurst 1996). This indicates that fires at 3-4 year intervals do not allow recovery of this vegetation type.

Weeds are not a major component of Box-Ironbark forest, with the exception of shaded, damper areas (Arthur Rylah Institute, unpublished data), and there may be little impact of fire on their abundance. For example, in dry sclerophyll forest north-east of Melbourne, applied burning led to minimal weed invasion (Adams and Simmons 1996). However, fire may affect the abundance of weeds in remnant vegetation patches, particularly if patches have a high perimeter:area ratio. Meredith (1987) suggested that weedy, cool-season annual grasses such as *Briza* (Quaking Grass) and *Bromus* (Brome) would be favoured by autumn burning. Similarly, in dry sclerophyll roadside vegetation in Western Australia, annual grasses such as *Ehrharta* (Veldt-grass) were favoured by burning, with effects still noticeable seven years later (Milberg and Lamont 1995), perhaps making that vegetation more fire-prone. A total of 18 weeds species were recorded in burnt roadside vegetation, of which eight had not been recorded before the fire (Milberg and Lamont 1995).

The results from the Wombat State Forest showed that there was no single fire regime that either favoured or disadvantaged all vegetation types (Tolhurst 2003), with long-rotation spring burning appearing to be the most benign in terms of not significantly disadvantaging any broad lifeform group. Nonetheless, reductions in the cover-abundance of different individual species were observed with each burning treatment, indicating that the maintenance of all species at a site under any proposed burning regime must involve a range of seasons and fire-frequencies.

However, while fire may be useful to maintain existing species diversity in Box-Ironbark forests, there appears to be little prospect of using fire as a management tool to enhance species richness. Orscheg (2006) surveyed the soil seed bank at 48 Box-Ironbark sites, and found that seed densities were low (only 55-91 germinants/m^2), and the composition represented a sub-set only of the extant vegetation community. Fire promoted short-lived obligate seeders in the first year post-burning, but these persisted in the extant community despite declining in abundance thereafter. Within two years the floristic composition in burnt sites was approaching that in unburnt sites, due largely to the reduction in short-lived obligate seeders and the high proportion of resprouters in the community, the abundance of which remained relatively stable (Orscheg 2006). Similarly, species diversity in burnt Box-Ironbark forest around Bendigo was similar to that in the unburnt controls, and relative differences in species abundance had largely disappeared after eight years (Hartskeerl 1997). Most importantly, burning did not replace 'lost' components of the system (Orscheg 2006).

Reseeder density in the Box-Ironbark seed-bank increased with less frequent fire, and with more time-since-fire (Orscheg 2006), suggesting that many seeds were relatively long-lived. Germination of twelve Box-Ironbark species studied was clearly enhanced by fire-related cues, but the overall seed-bank response suggested that while Box-Ironbark flora was tolerant of fire, and some elements flourished with it, no element was entirely reliant on it (Orscheg 2006). Animal-dispersed seeds were better represented in the soil seed-bank than wind-dispersed seed (Orscheg 2006), a pattern that is perhaps the opposite of what might be expected in small, isolated remnants (with less animals and more edge effects).

With a lack of undisturbed reference sites and limited historical records, we will never know with certainty the pre-European floristic structure of Box-Ironbark forests, or the extent to which we have lost certain fire-dependent or fire-sensitive plant species (Orscheg 2006). We can safely assume that the vegetation has changed, and we can certainly assume that the composition of remnants will be substantially different to that of larger tracts of forest. Management of these remnants, for the purposes of sustaining ecosystem processes and biodiversity, must be based not on an idealised concept of the pre-European condition, but on the likely effects of that management on the extant community in individual blocks. The response of remnants to fire will rely heavily on the composition of species currently growing on the site, and the nature of the soil seed bank (which in turn is heavily dependent on previous management and disturbance history). For large, relatively intact remnants with a known site history, it should be possible to provide some estimate of the likely response of the vegetation to an applied burn. However, for small sites, especially where they have had a long (probably undocumented) history of disturbance, any estimate of response would be purely speculative.

The response to fire management of remnant vegetation will vary based on complex relationships between fire frequency, season, intensity, and the life-cycle strategies of the individual plant species or seeds (their ‘Vital Attributes’, see ' *Plant Adaptations to Fire - a General Overview*'). Vital attributes include modes of establishment and persistence, and time to reach reproductive maturity and longevity, and can be used to model the effects of various fire regimes on plant composition (Noble and Slatyer 1981). Vital Attributes may thus be used to determine tolerable fire regimes for particular forest types that allow all species to be maintained in that forest type. For example, in Heathy Dry Forest (which has floristic similarities to parts of the Box-Ironbark ecosystem) the tolerable fire frequency range (floristics) is 6-50 years (Tolhurst 1999b). However, in any one patch the minimum inter-fire period for a fire that only kills the understorey (and that allows full recovery of structure) is 25 years (Tolhurst 1999b).

The draft management plans for the Castlemaine Diggings National Heritage Park (PV 2005) and Greater Bendigo National Park (PV 2006a), which contain Heathy Dry, Valley Grassy and Box-Ironbark Forests, state that they "*contain a high proportion of species that require fire for regeneration and are therefore vulnerable to long periods of fire exclusion*". However, there is no indication in those plans of what these species might be. Compilation of such data is therefore a priority, and while the draft Ecological Fire Strategy for Northern Bendigo (DSE 2006) makes an important start, there are still many gaps in fire response data.

For the purpose of this Literature Review, a list of Box-Ironbark vascular plant species was compiled and analysed, using available data (DSE and Parks Victoria) and expert knowledge, to determine those species that would be most informative for fire research. Criteria for appropriate species included their distribution, ease of identification, a known response to fire frequency, and reasonable certainty with regard to longevity of life stages, sensitivity to successive fires and conditions for establishment. Forty Key Fire Response species are presented in Table 2, while details of 156 common or widespread Box-Ironbark species are provided in Appendix 2.

Two species were considered most suitable as key response species (KFR = 1). *Xanthorrhoea glauca* subsp. *angustifolia* (Grey Grass-tree) is a long-lived perennial with a distinct growth form that resprouts after fire (Table 2). The depth of the skirt indicates time since last fire, and it has its highest density in infrequently burnt vegetation and its lowest seedling density in frequently burnt vegetation. *Brachyloma daphnoides* (Daphne Heath) is also a long-lived perennial, but with a seed-bank that germinates completely after fire. It requires ten years to reach reproductive maturity, then reaches its highest density at intermediate to low fire frequencies. Other suitable species (KFR = 2 or 3) had various characteristics, such as complete germination of seeds after fire (SD = C or G), death of juvenile resprouters (VG = W), inability to establish in older vegetation (TIR = I), lengthy juvenile phase or shorter time to local extinction. In all cases there is a known relationship between density (or cover) and time since fire.

The shortest tolerable inter-fire period in any one patch is determined by those species that take the longest time to reach reproductive maturity, but because these data are determined as the time to first seed set, the actual fire interval may need to be up to double the juvenile life (Tolhurst 1999b). Fires at less than 10-20 year intervals may thus lead to the local extinction of *Eucalyptus viridis* (Green Mallee), *Eucalyptus behriana* (Bull Mallee), *Brachyloma daphnoides*, *Exocarpos cupressiformis* (Cherry Ballart), *Grevillea alpina* (Cat's Claw Grevillea) and *Leptomeria aphylla* (Leafless Currant-bush) (Table 2 and Appendix 2), and no doubt other species (particularly woody ones) for which longevity data are still being compiled.

The longest tolerable inter-fire period in any one patch is determined by the species that take the shortest time to reach local extinction in the absence of fire. The data suggest that this is *Daviesia leptophylla* (Narrow-leaf Bitter-pea), after 20 years. However, this is likely to be an underestimate, because numerous individuals are still present in forest at Paddys Ranges near Maryborough 22 years after wildfire (Arthur Rylah Institute, unpublished data), and their seeds might be expected to be viable for some years yet. Of the other 103 species for which a species life was determined (Appendix 2), 24 are expected to persist for at least 20-50 years, and 79 should persist for at least 50-100+ years. Only around one quarter of these species is considered to be intolerant of germination in older vegetation.

These limited data suggest that Box-Ironbark species are adapted to long inter-fire periods, and most do not require regular fire to maintain biodiversity. However, they also suggest that most species will not be adversely affected by fires if they occur at not less than 10-20 year intervals.

We conclude this section on a precautionary note. The impact of successive fires becomes progressively greater (Tolhurst 2003), and their effects cannot be simply extrapolated from the effect of a single burn. In addition, management prescriptions developed at one site may not be suitable at another, even similar, site (Williams *et al.*

1994), because of the wide variations in fire behaviour and species response. Private remnants of Box-Ironbark forest may be small, and the nature of their soil seed banks and disturbance histories largely unknown. Size will be a major factor in species response to fire, by affecting initial species composition and the availability of propagules. For example, species richness was generally lower in smaller Grassy White Box Woodland remnants, particularly if grazed (Prober and Thiele 1995). Time since fragmentation may also be a factor, as the species richness of small sites in dry forest in New South Wales was significantly lower in older fragments compared to those more recently created (Ross *et al.* 2002). The effects of applied fire on individual remnants are thus likely to differ greatly, making the impacts on biodiversity at both a local and regional scale difficult to predict.

The justification for applied burning of Box-Ironbark remnants must be made on an individual site basis. This will be addressed further in Phase 2 of the project.

Key findings

- The response of the understorey to applied burning will depend heavily on the nature of individual remnants, season, landscape position, soil type, seed bank, disturbance history and susceptibility to edge effects.
- Some species, particularly leguminous shrubs and short-lived obligate seeders, will be promoted by a single fire in the short-term.
- Resprouting species that make up a large proportion of the flora will be little affected unless burning is frequent.
- Few species rely on fire for germination, and most species that are stimulated by fire will still recruit at a low level in the absence of fire.
- No species should be lost through burning provided the inter-fire period allows all species to reach reproductive maturity (a minimum of 10-20 years) but absent species are highly unlikely to reappear.
- Most species will persist even when the interval between fires exceeds 50 years.
- No single fire regime will either favour or disadvantage all vegetation types.
- Some individual species may benefit from occasional fire.
- The case for ecological burning of remnants must be made on an individual site basis.
- For many small remnants the response to burning can not be predicted.
- Further research is required into the germination requirements of Box-Ironbark shrub species, and the effects of applied burning (taking advantage of DSE's fuel-reduction program)

Key to Table 2 (opposite) and Appendix 2 (page 81)

BIB Ind. (fidelity to Box-Ironbark); R- Largely restricted to part of BIB; R Largely restricted to & widespread within BIB, rare elsewhere; C- Common in BIB & found in other vegetation, but not common; W Widespread in BIB & other vegetation; U Uncommon in BIB & rare elsewhere.

KFR (Suitability for estimation of fire history and monitoring); 1 = Highly Suitable, 2 = Suitable, 3 = Somewhat Suitable, NS = Not Suitable, - = not assessed.

SD (Mode of seed regeneration post-fire); D = Widely dispersed, seed available at all times after fire; S = Long-lived seed bank, seed stored, partial germination after fire; G = Long-lived seed bank, complete germination after fire; C = Short-lived seed bank, exhausted after single disturbance; Z = does not re-establish from seed immediately (or soon after) fire.

VG (Mode of vegetative regeneration post-fire); V = Sprouters, all ages survive, all become juvenile; U = Sprouters, mature remain mature, juvenile remain juvenile; W = Sprouters, mature remain mature, juveniles die; Y = Sprouters, mature become juvenile. juveniles die or there are no juveniles before the fire; X = does not resprout post-fire.

TIR (establishment conditions); T = can establish immediately after fire and throughout later years as well, thru to over-mature vegetation; I = can establish straight after fire, but not in older vegetation; R = cannot establish straight after fire but able to establish in older vegetation; M = can only establish in intermediate-aged vegetation, cannot establish straight after fire nor in over-mature vegetation; K = can establish immediately after fire and in over-mature vegetation, cannot establish in intermediate-aged vegetation.

JUV (time to onset of reproductive maturity, in years).

SPP LIFE (time to local extinction, including demise of seed store, in years); 3 = < 3 years, 10 = 3-10 years, 20 = 10-20 years, 50 = 20-50 years, 100 = > 50 years.

IND LIFE (lifespan/form of photosynthetic plants, excluding longer life of stored seed); A = Annual; ASP = Annual or short-lived perennial; SP = Short-lived perennial, < 10 years; MP = Medium-lived perennial, 10-50 years; LP = Long-lived perennial, > 50 years.

MAX DEN (fire frequency at which the density of a species is expected to be highest); L = low, I = intermediate, H = high.

FFG (Flora & Fauna Guarantee Act 1988); f = listed.

EPBC (The Environment Protection and Biodiversity Conservation Act 1999); v = vulnerable, E = endangered.

VROTS (Victorian Rare or Threatened Status); r = rare, v = vulnerable, e = endangered.

Table 2. Key Fire Response vascular plant species in Box-Ironbark forest. See key opposite.

Scientific Name	Common Name	BIB Ind.	KFR	SD	VG	TIR	JUV	SPP LIFE	IND LIFE	MAX DEN
Xanthorrhoea glauca subsp. *angustifolia*	Grey Grass-tree	R	1		U	T		100	LP	L
Brachyloma daphnoides	Daphne Heath	W	1	G	W	T	10	100	LP	I-L
Daviesia leptophylla	Narrow-leaf Bitter-pea	W	2			I	5	20	MP	I
Phebalium festivum	Dainty Phebalium	R-	2	G	X	I	5	50	MP	I
Acacia genistifolia	Spreading Wattle	W	2	S		I	5	50	MP	I
Acrotriche serrulata	Honey-pots	W	2		Y	T	5	50	LP	I-L
Dillwynia cinerascens s.l.	Grey Parrot-pea	W	2	S		I	5	50	MP	I
Dillwynia sericea	Showy Parrot-pea	W	2			I	5	50	MP	I
Gompholobium huegelii	Common Wedge-pea	W	2		X	I	5	50		H-I
Hakea decurrens subsp. *physocarpa*	Bushy Needlewood	W	2	G	X	I	5	50	LP	I
Persoonia rigida	Hairy Geebung	C-	2	G		T	5	100		L
Eucalyptus polybractea	Blue Mallee	R-	2	G	Y	I		100	LP	H-I
Amyema miquelii	Box Mistletoe	W	2	D		R	5	100	MP	L
Calytrix tetragona	Common Fringe-myrtle	W	2	G	V	T	5	100	MP	I-L
Cassinia arcuata	Drooping Cassinia	W	2	S		T	5	100	MP	I
Goodenia geniculata	Bent Goodenia	W	2	G	V	I	2	100	MP	I
Joycea pallida	Silvertop Wallaby-grass	W	2	Z	V	T	2	100	LP	L
Themeda triandra	Kangaroo Grass	W	2	C	Y	T	2	100	LP	L
Swainsona behriana	Southern Swainson-pea	U	2	Z	W	I	1		MP	I-L
Daviesia ulicifolia	Gorse Bitter-pea	W	2		Y					I
Acacia difformis	Drooping Wattle	C-	3	S		I	5	50	MP	I
Prostanthera denticulata	Rough Mint-bush	R	3			I	5	50		I
Pultenaea largiflorens	Twiggy Bush-pea	R	3			I	5	50		I
Astroloma humifusum	Cranberry Heath	W	3	G	W	T	5	50	MP	I-L
Chrysocephalum apiculatum s.l.	Common Everlasting	W	3		V	T	2	50	ASP	H
Eutaxia microphylla var. *diffusa*	Spreading Eutaxia	W	3			K	5	50		I
Exocarpos cupressiformis	Cherry Ballart	W	3	G	Y	T	10	50	LP	I-L
Hibbertia riparia	Erect Guinea-flower	W	3	G	V	I	5	50	MP	I
Hibbertia sericea s.l.	Silky Guinea-flower	W	3	G	Y	I	5	50	MP	I
Hypoxis glabella var. *glabella*	Tiny Star	W	3	S	U	I	2	50	MP	H
Pultenaea laxiflora	Loose-flower Bush-pea	W	3	S		I	5	50		I
Eucalyptus viridis	Green Mallee	R-	3	G	Y	I	10	100	LP	I
Eucalyptus behriana	Bull Mallee	W	3	G	Y	I/T	10	100	LP	H-I
Leptospermum myrsinoides	Heath Tea-tree	W	3		Y	I	5	100	MP	I
Pultenaea prostrata	Silky Bush-pea	W	3	S		I	5	100		I-L
Tricoryne elatior	Yellow Rush-lily	W	3		V	I	2	100		H
Bursaria spinosa subsp. *lasiophylla*	Hairy Bursaria	C-	3		V	T				L
Pultenaea graveolens	Scented Bush-pea	U	3			I			MP	I
Burchardia umbellata	Milkmaids	W	3		U					I-L
Correa reflexa	Common Correa	W	3	G	Y	T	5		MP	I

Effects of Fire on Canopy Species

The effects of fire on canopy structure depend substantially on the life history attributes of the main canopy species and the intensity of the fire. Many *Eucalyptus* species resprout after fire from epicormic buds under fire-resistant bark, and mature plants thus remain mature. Other species such as Mallee eucalypts or Snow Gum (*E. pauciflora*) resprout from underground lignotubers with the aerial stems often being killed. Other species utilise the post-fire nutrient flush and release from competition to germinate and establish the next generation [stand replacement, as in *E. regnans* (Mountain Ash)].

Box and Ironbark trees are considered to be fire resistant (Meredith 1987), and most will recoppice or resprout readily from epicormic buds after fire. In Australia, eucalypt canopies are persistent but often open, with more light reaching the understorey than in many other forest types, greatly reducing the need for understorey species to be particularly shade-tolerant (Kirkpatrick 1997). This might also favour overstorey recruitment in the absence of fire, and in Box-Ironbark forest there is generally a low level of constant seedling recruitment, particularly of *E. polyanthemos* (Red Box), *E. goniocalyx* (Long-leaf Box) and *E. macrorhyncha* (Red Stringybark) (Arthur Rylah Institute, unpublished data). Recruits of *E. tricarpa* (Red Ironbark) tend to be less abundant, and this is not surprising given that flowering and seed production in this species tend to be sporadic (Kellas 1991). Box-Ironbark forests may therefore not *need* recurrent fire to persist. For example, eucalypt recruitment was higher in unburnt dry sclerophyll forest than in lightly burnt forest (Purdie 1977b). However, further research is required to determine the germination requirements for *E. tricarpa*, as seeds of this species seem to be short-lived and lack dormancy, and while they germinate readily under laboratory conditions, they do not germinate readily under field conditions (Orscheg 2006).

The response of canopy species to fire may vary substantially depending on factors such as the season of burn or fire intensity, and pre- and post-fire conditions. For example, in Cumberland Plain Woodland, a single burn in summer, but not winter/spring, led to an increase in the richness and abundance of eucalypts compared to that in the unburnt treatment (Hill and French 2004). Much of the effect of fire intensity relates to the amount of seed fall stimulated, hence post-fire seedling abundance. In dry sclerophyll forest near Canberra, eucalypt seedlings were more abundant in the burnt treatment only when the fire was of sufficient intensity to scorch the canopy (Purdie 1977b). Similarly, in the Wombat State Forest, autumn burning with a greater degree of canopy scorch led to higher cover of eucalypt recruits after about four years (Tolhurst 2003).

There is some evidence from early historical accounts that increased frequency of fire, particularly hot fire, was already changing the canopy composition of some Gippsland forests before 1900 (Chesterfield 1984). For example, young Stringybark scrub was reported to be crowding out Box eucalypts around Lower Tambo and Lake Tyers, and crowding out *E. tricarpa* north and north-west of Bruthen (Anon. 1900, cited in Chesterfield 1984). Similarly, *E. Muelleriana* (Yellow Stringybark) was replacing *E. sieberi* within frequently burnt areas leased for grazing (Incoll 1940, cited in Chesterfield 1984).

Coppice growth after fire can have a marked impact on overstorey structure, by changing single-stemmed individuals into multi-stemmed individuals. For example, frequent historical burning of Peppermint forests in eastern Victoria encouraged the formation of dense coppice thickets of *E. dives* (Broad-leaved Peppermint), particularly on ridge tops (Wakefield 1970). There may be similar implications for Box-Ironbark forest, given that most *Eucalyptus* species present have a strong capacity for recoppicing.

The tolerable fire frequency range that allows all species to persist is 6-50 years for Heathy Dry Forest (Tolhurst 1999b). In any one patch the minimum inter-fire period for a fire that kills or severely reduces the overstorey and that allows full recovery of structure is 60 years (Tolhurst 1999b). However, fires of sufficient intensity and size to kill large areas of overstorey in Box-Ironbark forest are likely to be relatively infrequent. For example, only around 15 of the 800 wildfires that were recorded on public land across the Box-Ironbark region between 1980/81 and 1995/96 exceeded 100 ha in size: around 80% were less than 5 ha (ECC 1997). Few of these fires were caused by natural ignition, and their relatively small size would suggest that fire intensity would be low.

The occurrence of large or high-intensity fires on smaller fragmented remnants might be expected to be substantially less than in larger expanses of forest, with the possible exception of remnants where management activities or disturbance have led to the development of dense shrub cover. Thus, fire cannot be relied on to facilitate the renewal of the canopy in Box-Ironbark remnants. In any event, given the low-level, continual recruitment of most overstorey trees in Box-Ironbark forests, stand senescence is unlikely to be of concern, and fire for the purpose of stand replacement might be seen as largely redundant.

Key findings

- The response of the canopy will depend on the intensity of the fire. In most instances, fire in small remnants is unlikely to be of sufficient intensity to lead to canopy replacement.
- Most Box-Ironbark canopy species show continual recruitment in the absence of fire, and thus do not rely on it. Fire for the purpose of stand replacement might be seen as largely redundant.
- In any one patch, the minimum inter-fire period for a fire that kills or severely reduces the overstorey and that allows full recovery of structure is likely to be around 60 years.
- Further research is required to determine the germination requirements for Ironbark eucalypts.

Effects of Fire on Litter Accumulation and Fuel

Litter is an important part of the forest habitat for many species of invertebrates, small mammals, birds, reptiles and fungi (Tolhurst and Kelly 2003). However, dead leaves, bark and twigs (that make up fine fuel) are also important in the propagation of fire and flame height (Tolhurst and Kelly 2003).

In the Wombat State Forest, a mixed foothill forest south of the Box-Ironbark study area, the litter (fine fuel) load varied from around 9.4 to 26.4 t/ha, with an average steady state load of around 16 t/ha (Tolhurst and Kelly 2003). Litter accumulation tended to peak in the summer-early autumn period, such that the surface fine fuel load on the forest floor could be substantially higher at the end of the fire season than it was at the beginning (Tolhurst and Kelly 2003). Unburnt Box-Ironbark forest around Bendigo had average fuel loads substantially lower than this, ranging from 4.3 to 6.9 t/ha (Hartskeerl 1997). Surface fine fuel load was also lower in *E. sideroxylon* - *E. macrorhyncha* forest within Chiltern Regional Park, ranging from 6.0 to 8.0 t/ha, with average fuel depth of 2.1 to 2.9 cm (Chatto 1996). Similarly, data from the Box-Ironbark ecological thinning trial (a Parks Victoria experimental management project) showed that the cover of leaf and fine litter ranged from around 60-70%, while mean litter depth ranged from 1.7 to 2.9 cm in Box-Ironbark Forest, and 1.8 to 3.5 cm in Heathy Dry Forest (University of Ballarat, unpublished data). This supports claims that the overall rate of fine fuel accumulation in Box-Ironbark forest is relatively low (Kellas 1991). Nonetheless, fine fuel loads of 2-3 t/ha, covering less than 60% of the ground surface, may carry fire if the fuel is dry, and if wind speeds are sufficient (Tolhurst and Cheney 1999).

Litter decomposition also varies substantially depending on season and time since fire (Tolhurst and Kelly 2003). Following fire, this decomposition is disrupted due to reductions in decomposers, and an environment that is not conducive to microbial or arthropod activity (Tolhurst and Kelly 2003). Litter fuels in the Wombat State Forest were reduced by around 40-80% after experimental fires, but approached equilibrium load by around four years post-fire (Tolhurst and Kelly 2003). There was no evidence from these experiments that the rate of litter reaccumulation differed after short rotation (3 years) or long rotation (10 years) burning (Tolhurst and Kelly 2003). The mean fuel load in burnt Box-Ironbark forest was 7 to 50% lower than in unburnt forest after one year, and remained 30% lower after four and eight years (Hartskeerl 1997). This study was hampered by a lack of replication, but nonetheless it suggests that litter accumulation post-fire in Box-Ironbark may be relatively slow compared to that in other forest types.

The structure of a forest stand and the density of the canopy are major factors in the rate and amount of fuel accumulation (Tolhurst and Kelly 2003). Box-Ironbark forest with structural characteristics reminiscent of the presumed pre-European condition (sparse undergrowth and open nature) appears unlikely to be prone to high fuel accumulation, supporting assertions by Kellas (1991) that natural fire hazard is low compared to other forests.

Litter is also important in the nutrient cycle in these forests, because the surface layer (0-2 cm) of the soil on the forest floor includes a relatively large proportion of partly decomposed litter and coarse organic matter (Hopmans 2003). During experimental fires in the Wombat State Forest, this material was readily volatilised, and burning at three-year intervals led to a reduction in carbon and nitrogen in the surface soil (Hopmans 2003). The changes were greater at sites where the fire was more intense, leading the author to suggest that frequent burning would lead to a decline in soil organic matter and fertility. Similarly, repeated burning of wetter *E. obliqua* forest at 5-7 year intervals is expected to result in losses of nitrogen from the ecosystem (Hamilton *et al.* 1991).

Key findings

- Litter plays an important role in nutrient cycling, and provides important habitat for invertebrates and small vertebrates.
- Litter accumulation is relatively low compared to that in other forests, but it may still carry fire.
- Frequent burning (for example, 3-5 year intervals) will disrupt natural processes in the short-term and may eventually lead to a depletion of soil nutrients.

Effects of Fire on Invertebrates

Invertebrates are an often overlooked component of forest ecosystems, yet they constitute 95% of known fauna species in Australia (York 1996) and play a vital role in biological control, nutrient recycling, decomposition of organic matter and aeration of the soil (Collett and Neumann 2003; Leonard 1974). As such, the impact of fire on many invertebrates is intimately related to the effect of fire on the litter layer and soil (Leonard 1972; Suckling and Macfarlane 1984). However, despite their ecological importance, there have been few detailed studies of the effects of fire on this faunal group (Collett and Neumann 2003; Friend *et al.* 1999; Meredith 1987).

The high diversity of invertebrates in forest ecosystems results in a variety of different responses to fire depending upon the season of the fire, fire intensity (which determines which micro-environments act as refuges for invertebrates), the extent of fire (which affects recolonisation), and the frequency of fires. One of the consequences of these factors is that attempts to study the effects of fire on invertebrates often give conflicting or confusing results. The natural variability associated with invertebrates (in composition, activity periods, microhabitat use, environmental factors etc.) can sometimes mask the effects of fire and make responses difficult to predict (Campbell and Tanton 1981; Friend 1996). Further, it may be impossible to identify the climax community or baseline state with which post-fire populations can be compared, or to which management should aim (Friend 1996).

In Australia there have been studies on fire effects on invertebrates in wet eucalypt forests of Victoria and Western Australia, Mallee vegetation in Victoria, savanna country in the Northern Territory, and in the arid zone (Northern Territory). The studies that are more relevant to Victoria's Box-Ironbark forests are the studies in the drier eucalypt forests of New South Wales (Andrew *et al.* 2000; Campbell and Tanton 1981; Meredith 1987; York 1999, 2000), Queensland (Vanderwoude *et al.* 1997), Victoria (Collett 2003; Collett 1998, 1999; Collett and Neumann 1995; Collett *et al.* 1993; Leonard 1972; Neumann *et al.* 1995) and Western Australia (Friend 1995; Majer 1985; Van Heurck and Abbott 2003). Comparison of these invertebrate fire studies in the drier forests does not provide general conclusions because each study used different techniques and worked on different invertebrate groups.

The Victorian studies were conducted in dry sclerophyll eucalypt forest in the Wombat State Forest near Daylesford. They involved prescribed low intensity fires at the same sites in 1985 and 1988 (Collett and Neumann 1995; Collett *et al.* 1993) with an additional fire in 1991 (Collett 2003; Collett 1999). Pitfall traps (18 mm diameter test tubes) were the primary collecting method for ground active invertebrate orders (Collett *et al.* 1993) and beetles identified to morphospecies (Collett and Neumann 1995), and the litter/upper soil was searched for earthworms (Collett *et al.* 1993). Pitfall traps were used in the 1991 study and invertebrates were identified to order (Collett 1999).

Collett *et al.* (1993) concluded that the 1985 and 1988 fires had no discernible cumulative effect on the total numbers of arthropods, non-insects and arthropod decomposers, though total predators increased significantly after the second fire due largely to earwigs (Dermaptera). Springtails (Collembola) declined for 12 months after the first fire, and earthworms also declined and took three years to recover. Collett *et al.* (1993) suggested that two consecutive fires may disrupt the decomposer cycle for up to three years because of the effects on springtails and earthworms. The effects of fire on beetles (Coleoptera) and flies (Diptera) were difficult to interpret and their numbers may have been more influenced by seasonal factors. Collett *et al.* (1993) suggested that any spring prescribed fires should only be applied at intervals of more than three years to minimize adverse effects on invertebrates, and burning should not be undertaken during droughts.

Collett and Neumann (1995) identified 31 families and 105 species of litter-dwelling beetles. The major families were the predominantly predatory Staphylinidae and the decomposer/fungus feeding Nitidulidae and Leiodidae. The two fires had no discernible effect on most of the beetles except for the Staphylinidae, which declined significantly after the second fire. The third fire had little discernible effects on most invertebrates except for a significant decrease in beetles and a significant increase in flies (this was also the trend in the controls) (Collett 1999). In Mallee vegetation, the number of beetle species was higher in more recently burnt sites, although the total abundance did not change between fire treatments (Schlesinger *et al.* 1997).

Collett (2003) assessed invertebrate orders collected by pitfall trap over a 14 year period (1985-1999) with regard to prescribed autumn and spring fires. He could not determine whether the changes at the ordinal level were due to fire or non-fire factors, and suggested that further study at the species level was required.

Most of these studies would have benefited from higher levels of pre-fire sampling (to quantify natural variation), more standardised trapping methods and identification to lower taxonomic levels (Friend 1995). York (1999) found that frequent burning caused significant decreases in abundances of Acarina, insect larvae, Diptera and Coleoptera, probably because these groups are dependent upon leaf litter, while significant increases occurred in Hemiptera, ants and spiders, possibly due to changes in the environment and also increased ease of trapping. With ants, beetles, flies, spiders and bugs, overall species richness did not change with frequent burning, but there was a loss of litter layer dependent species (stable moist conditions) but these were replaced by species with broader tolerances. There were substantial shifts in species composition, and York (1999) suggests that frequent and extensive fuel-reduction burning could result in a reduced invertebrate biodiversity at the regional scale. Andrew *et al.* (2000) highlighted the importance of preserving a range of microhabitats (such as logs) to conserve invertebrate biodiversity in areas subjected to frequent low-intensity burns.

Ants have been used as an indicator of fire effects (Andersen and Yen 1985; York 1996, 2000). However, some care has to be taken with using ant species richness, as response to fire can vary substantially. Andersen and Yen (1985) found that ant activity and richness increased after a fire in Mallee vegetation, as a result of simplification of the habitat and release from inter-specific competition. In contrast, Andrew *et al.* (2000) found that regular low-intensity burning of forests in New South Wales did not impact on ant diversity even though they were structurally simplifying the vegetation. York (1996) found that burnt and unburnt sites in dry sclerophyll forest (New South Wales mid north coast) contained distinctive communities with particular species associations. Burnt areas were characterised by ant species with broad dietary requirements and environmental tolerances, with a loss of rare or highly specialised species (York 1996), therefore it is important that some other criterion, such as the structure of ant functional groups that considers groups of ants that rapidly colonise disturbed areas, be used.

With regard to the Box-Ironbark forests, some general guidelines can be suggested to avoid adverse impacts on invertebrates. Burning should be conducted at a minimum of three year intervals, and burning in times of extended drought should be avoided. Private remnants are highly unlikely to be burnt at such a high frequency, therefore occasional burning is not expected to have any long-lasting adverse effects. Nonetheless, if any burning is undertaken, special microhabitats (such as logs, large accumulations of woody debris or low lying areas with wetter litter) should be left unburnt where possible. Further research is required to determine what effect the reduction by fire of coarse woody debris has upon invertebrates, particularly termites, which rely upon this debris as habitat and as food.

Key findings

- Research into the effects of fire on invertebrates is confounded by high natural variability, and it is often difficult to determine the baseline or climax community.
- Short-term effects are intimately linked in many cases to burning of the litter layer, and burning at frequencies as low as every three years should be avoided.
- More research is required into the effects of fire on termites, known to be key drivers of secondary productivity in these forests.

Effects of Fire on Birds

Fire intensity plays both a direct and indirect role in bird mortality and persistence. Few birds are directly killed in low-intensity fires (Suckling and Macfarlane 1984), because most are highly mobile, but many may be killed when the fire is very intense (Brown *et al.* 1998; Wilson 1996; Woinarski and Recher 1997). For example, following the Ash Wednesday fires near Lorne, over a thousand burnt bird carcasses were recorded along 49 km of coastline (Wegener 1984), representing only a small sample of the total number killed. Similarly, on Ash Wednesday, small colonies of the endangered Helmeted Honeyeater (*Lichenostomus melanops cassidix*) were lost after intense fire at Cockatoo and Cardinia Creeks (Backhouse 1987).

The indirect effects of fire, by removing food and habitat resources, will then vary substantially depending on birds' trophic guilds and the vegetation strata affected (Wilson 1996). The post-fire trend in bird abundance is usually characterised by an immediate, abrupt decline, followed by a period of recovery (Green and Sanecki 2006; Woinarski and Recher 1997). For example, three weeks after the Lerderderg Gorge fire (Wombat State Forest), previously common species such as Golden Whistler (*Pachycephala pectoralis*), Rufous Whistler (*Pachycephala rufiventris*), Grey Fantail (*Rhipidura fuliginosa*), Honeyeaters and Pardalotes were virtually absent in the burnt areas (Hewish 1983). However, post-fire composition depends largely on the mobility of individual species and resource availability, and initial responses do not necessarily predict longer-term responses (Loyn 1997). For example, Laughing Kookaburras (*Dacelo novaeguineae*) have been recorded in high numbers after fires taking advantage of both fire-related carrion and the reduced cover for prey items (Woinarski 1999). Responses can also involve seasonal migration patterns.

The decline and subsequent recovery of bird communities after fire will vary substantially between vegetation types, and may not be consistent between studies. For example, in far eastern Victoria, bird abundance recovered rapidly in rainforest but slowly in heathland, the opposite of what would be expected given the presumed adaptations to fire by the respective vegetation types (Loyn 1997). Generalisations about fire adaptations may not help to predict fire response (Loyn 1997), creating difficulty in extrapolating research results from different forest types to Box-Ironbark forest.

One study of relevance to Box-Ironbark forest is likely to be that from the fire effects study in the Wombat State Forest, which examined effects of fuel reduction burning in a slightly wetter forest type. The forest had local occurrences of bird species more typical of the drier Box-Ironbark forests (Loyn *et al.* 2003), and areas of similar vegetation structure. One year after burning, little difference in species composition was found between burnt and long-unburnt treatments (Loyn *et al.* 2003). Open-ground foragers, including Scarlet Robin (*Petroica multicolor*), Flame Robin (*Petroica phoenicea*) and White-winged Chough (*Corcorax melanorhamphos*) were more common on burnt areas, particularly when burnt in autumn, although the seasonal differences were not significant (Loyn *et al.* 2003). Eastern Yellow Robin (*Eopsaltria australis*) and Superb Fairy-wren (*Malurus cyaneus*) were abundant in burnt areas three weeks after wildfire in the Lerderderg Gorge (Wombat State Forest) in 1983 (Hewish 1983). The abundance of open-ground foragers was significantly higher in frequently burnt areas, while the abundance of large hollow-nesters was also higher (but not significantly so) (Loyn *et al.* 2003). There was weak evidence that the abundance of shrub foragers was reduced in frequently burnt areas (Loyn *et al.* 2003). Interactions were noted between burning season and frequency in the Wombat State Forest research. Open-ground foragers were most numerous in sites that were frequently burnt in autumn, and least numerous in sites that were infrequently burnt in autumn (Loyn *et al.* 2003). Damp-ground foragers, including Eastern Yellow Robin and Grey Currawong (*Strepera versicolor*), were adversely affected by both infrequent autumn burns, and frequent spring burns (Loyn *et al.* 2003). Low-understorey foragers, such as the White-browed Scrubwren (*Sericornis frontalis*), were least numerous after frequent spring burning (Loyn *et al.* 2003), but not so affected by frequent autumn burning (although the interaction was not significant).

Low-intensity fuel reduction burns, by consuming only the lower strata, may affect mostly ground-dwelling and understorey birds in the year after the fire (Christensen and Kimber 1975), while the removal of nectar-producing plants such as *Banksia* will have an immediate effect on honeyeaters (Meredith 1987). Burning of deep litter is likely to temporarily affect species such as Painted Button-quail (*Turnix varia*) and Spotted Quail-thrush (*Cinclosoma punctatum*), while burning of logs may affect species such as the Brown Treecreeper (*Climacteris picumnus*) (Andrew Bennett, Deakin University, pers. comm.). However, intense wildfire will drastically affect birds from all levels of the forest (Loyn 1997; Meredith 1987; Suckling and Macfarlane 1984), and for a longer period of time. Following wildfire in the Snowy Mountains, few bird species remained in the burnt areas, and the post-winter influx of migrants was significantly reduced (Green and Sanecki 2006).

The effects of canopy fires on birds may be also gradual and indirect. In some forests, such as in East Gippsland, *Symphyomyrtus* eucalypts tend to harbour more leaf-gleaning birds such as Thornbills and Pardalotes than do *Monocalyptus* sub-genera, due to their greater attractiveness to invertebrate species (Woinarski and Cullen 1984). Changes in the relative abundance of eucalypt sub-genera through inappropriate burning regimes may thus lead to changes in the abundance of those leaf-gleaning birds.

Following a single fire, most Australian vegetation tends to revert back to its previous structure and composition over time, and bird communities should also recover to pre-fire composition (Meredith 1987). Indeed, birds appear to be surprisingly resilient to the transitory effects of fire (Christensen and Kimber 1975; Smith 1989). The effects of fuel reduction burning on birds also appear to be far milder than the effects of severe wildfire, particularly on nectarivores and canopy foragers (Loyn *et al.* 2003; Woinarski and Recher 1997). Fires that promote low shrub growth may actually advantage some species, such as Crested Bellbird (*Oreoica gutturalis*), Gilbert's Whistler (*Pachycephala inornata*), Fairy-wrens and Brown Thornbills (*Acanthiza pusilla)*, while promotion of *Acacia pycnantha* is likely to benefit the Swift Parrot (*Lathamus discolor*) (Mac Nally and Horrocks 1999). However, fires that are too frequent to allow the plants in various strata to reach reproductive maturity may lead to a long-term shift in vegetation structure, and bird species that require mature vegetation are likely to be lost (Meredith 1987; Wilson 1996). These species are often specialists with restricted distribution (Meredith 1987), and most threatened species for which research is available show a clear preference for less frequent fire

(Woinarski and Recher 1997). For example, some Honeyeaters require dense, closed shrub cover, hence relatively long inter-fire periods, and they will be absent for extended periods after intense fires (Woinarski and Recher 1997). Similarly, in the Mallee, some bird species are restricted to vegetation older than 20 years (Meredith 1984). Detailed long-term studies in Australia suggest that frequent low-intensity fires may lead to the decline and loss of bird species currently considered common (Woinarski and Recher 1997). However, total fire exclusion might also be undesirable in some instances, as birds such as White-winged Chough, Spotted Quail-thrush, Blue-winged Parrot (*Neophema chrysostoma*) and Red-browed Finch (*Neochmia temporalis*) respond positively to fire in some circumstances (Loyn *et al.* 2003).

Despite the resilience to fire shown by many birds, and despite the assumed resilience of vegetation in some fire-prone ecosystems, bird populations may nonetheless be severely disadvantaged by fires that are too frequent or too large. For example, in a heathland study site in Western Australia, fire occurs on average once every two years (Brooker 1998), and could be considered a natural event to which fauna species would be adapted. However, a major fire in 1985, followed rapidly by two smaller reburns, led to substantial declines over the next eight years in the numbers of Splendid Fairy-wrens (*Malurus splendens*), Western Thornbills (*Acanthiza inornata*), Scarlet Robins and Yellow-rumped Thornbills (*Acanthiza chrysorrhoa*) (Brooker 1998). The author suggested that population recovery was affected by the extent of the area burnt in 1985, with all habitat burnt both inside and outside the study area, a result that would not have been predicted from previous studies. It is clear that fire management based on limited data could yield recommendations that are detrimental to the values that land managers are trying to promote (Brooker 1998).

Consideration must also be given to the landscape position and size of remnants. Gullies in Box-Ironbark forest tend to be more open, with more large trees, higher canopy cover and higher grass cover than ridges, and may more closely approximate the presumed pre-European condition (Mac Nally *et al.* 2000b). There is a substantial difference in species richness and abundance between ridges and gullies (Mac Nally *et al.* 2000b), and gullies therefore have a disproportionate importance in terms of both fauna richness and conservation value. Gullies may also act as a major refuge for birds in the event of a fire (Loyn 1997; Smith 1989), and draw in birds avoiding the burnt ridges after fire (Smith 1989). Some birds are gully 'specialists': in Box-ironbark they include Brown Treecreeper, White-plumed Honeyeater (*Lichenostomus penicillatus*), Red-rumped Parrot (*Psephotus haematonotus*), Willie Wagtail, Galah (*Cacatua roseicapilla*) and Crested Shrike-tit (*Falcunculus frontatus*) (Mac Nally *et al.* 2000b). Therefore, gullies with an avifauna distinct from that on ridges (Mac Nally *et al.* 2000b) must be managed separately to those ridges. There is also a difference in species richness and abundance between large and small Box-Ironbark remnants (Bennett 1999; Mac Nally *et al.* 2000a), and small remnants are unlikely to be able to retain environmental diversity or sustain populations of species associated with particular seral stages (Woinarski and Recher 1997). Thus, the results of research in large forest blocks may not necessarily be extrapolated to remnants.

Assuming that private remnants of Box-Ironbark forest are mostly small, and that rates of accumulation of litter and flammable vegetation (hence fuel hazard levels) are relatively low (Calder *et al.* 1994), then it is reasonable to assume that any ecological burning is likely to be of a limited size and of low intensity. The forest canopy is unlikely to be badly affected by low-intensity burning, and in similar forests this is where many of the commonest birds feed (Loyn *et al.* 2003). Thus, the effects of that ecological burning on birds are likely to be related to the effects on the structure and composition of the lower to middle plant strata. However, these lower strata may differ substantially between remnants with different (perhaps unknown) management and disturbance histories. For example, disturbed Box-Ironbark remnants often have few ground-foraging birds that rely on low shelter (Bennett 1999), and more common, open-country foragers. Predicting the effect of burning on bird communities in private Box-Ironbark remnants thus remains problematic, and predictions will need to take into account the nature of the remnant itself, and the likely response of individual birds found there.

Of the 213 bird species that have been recorded in the broader Box-Ironbark study area (ECC 2001), expert opinion suggested that 154 were likely to be in and around the forests of interest for this report. Expert opinion was further sought to gauge the impact (positive or negative, long-term or short-term) of fire on each of these species. Given the focus on applied ecological burning, and the relatively small size of many remnants, the assumption was made that fire would be of low-moderate intensity. Species that are expected to respond strongly are shown in Table 3, while the full species list is provided in Appendix 3.

Four birds (Emu, Brown Quail, Little Button-quail, Painted Button-quail) are expected to show a highly positive response in the long-term to both spring and autumn burning (Table 3). These birds are predominantly seed-eaters, and will benefit from increased seed production as a consequence of new herb and grass growth. However, as ground-nesters, it is predicted that they will be slightly disadvantaged by a spring burn in the short-term.

Table 3. Bird species likely to be most affected by a low-moderate intensity fire in Box-Ironbark forest. HP = highly positive, SP = slightly positive, HN = highly negative, SN = slightly negative.

Common name	Scientific name	Spring burn, short-term	Spring burn, long-term	Autumn burn, short-term	Autumn burn, long-term
Emu	*Dromaius novaehollandiae*	SN	HP	SP	HP
Brown Quail	*Coturnix ypsilophora*	SN	HP	SP	HP
Little Button-quail	*Turnix velox*	SN	HP	SP	HP
Painted Button-quail	*Turnix varia*	SN	HP	SP	HP
Bush Stone-curlew	*Burhinus grallarius*	HN	SN	SN	SN
Brown Treecreeper	*Climacteris picumnus*	HN	SN	HN	SN
White-winged Chough	*Corcorax melanorhamphos*	HN	SP	HN	SP

Three species are likely to be highly negatively affected in the short term by fire (Table 3). The Bush Stone-curlew is a threatened species (DSE 2003a) that nests, roosts and feeds on the ground, where it benefits from accumulation of coarse woody debris. It would be severely affected by loss of cover after a spring burn. The 'near threatened' Brown Treecreeper (DSE 2003a) is a ground-forager that would be affected in the short- or medium-term by the reduction in litter from burning in either season. Similarly, the White-winged Chough is exclusively a scavenging and ground-feeding species, and could be highly negatively affected immediately after a burn in either season [although data from the Wombat State Forest experiment showed little difference one year after burning (Loyn *et al.* 2003)].

Other Box-Ironbark birds are expected to show a wide range of responses to fire, depending on their nesting and foraging behaviour (Appendix 3). Around one third of species are likely to be slightly negatively affected in the short-term by a spring burn, and these are generally birds that rely on the ground or low foliage for nesting and/or foraging. Those that forage on the ground or in low foliage will also be slighty negatively affected in the short-term by an autumn burn, although many will benefit slightly in the long-term from the fresh growth promoted by fire. Only ten species are expected to be slightly negatively affected in the long-term by low-intensity fire in either spring or autumn, and these are generally birds that use the litter layer, such as Thornbills and Scarlet Robin. No species are expected to suffer highly negative effects in the long-term.

Raptors and parrots are well-represented within the 32 species that are considered likely to be slightly positively affected by a spring burn, and these may also be slightly positively affected by an autumn burn (Appendix 3), due to either greater exposure of prey animals, or the stimulation of flowering and seed production in many plant species. Around 45% of bird species may benefit slightly in the long-term from a fire, commensurate with any improvement in the structure and diversity of the vegetation and increases in flowering and seed production, even though they may be slightly disadvantaged in the short-term. Another 45% of species are expected to have a neutral response in the long-term.

In general, it was considered that spring burning could have a slightly greater negative impact on bird communities than autumn burning, although 90% of species were expected to respond similarly to a burn in either season (Appendix 3). In all cases, the long-term responses by birds to spring or autumn burning are predicted to be the same.

Key findings

- The effects of fire on birds are strongly dependent on the severity of the fire and the structural components of the forest that are burnt, and recovery of populations is linked to the recovery of the vegetation.
- Some seed-eaters may be advantaged in the long-term if fire promotes new vegetation growth, as may be species that take short-term advantage of post-fire carrion or the reduction in protective cover for prey.
- Species that rely on the ground layers for nesting or foraging may be negatively affected in the short-term by low-intensity fire, particularly if it interferes with breeding.
- The minimum inter-fire period is likely to be similar to that which will allow full recovery of understorey structure (i.e. at least 25 years).
- The burning of gullies, which have higher plant and animal diversity, should be avoided.

Effects of Fire on Mammals

Mammals tend to have complex but predictable relationships with vegetation and disturbance, with varying reliance on structural and floristic factors, including hollows (Irvin *et al.* 2003b). In general, the composition of the ground-dwelling mammal fauna appears to be dependent on the complexity of the understorey, and a decrease in understorey complexity may lead to a reduction in the abundance of small-medium mammals but an increase in the number of large mammals such as kangaroos (Catling and Burt 1995). However, the wide ranges of mammals, habitats and forest types make generalisations about response to fire difficult. Fire response can also differ markedly, depending on factors such as aspect, season, pre- and post-fire conditions, fire intensity and previous site history (Penn *et al.* 2003). Post-fire predation, such as by foxes, can also have a significant influence (Green and Sanecki 2006; Heislers 1974; Recher *et al.* 1974). Therefore, it is not surprising that there has been little use of fire specifically for management of native mammals (Wilson and Friend 1999).

The succession of small ground-dwelling mammals after fire is closely tied to vegetation succession because of the animals' requirement for food, shelter and breeding conditions (Friend *et al.* 1999; Friend 1993; Meredith 1987; Monamy and Fox 2000; Recher *et al.* 1974). For example, the Dusky Antechinus (*Antechinus swainsonii*) frequents logs and dense cover where litter is well-developed, forages on the ground during the day, and catches its insect prey from the litter and low vegetation (Recher *et al.* 1974). This is precisely the habitat that is burnt in even "cool" fires. Thus, many small mammals decline in abundance in the wake of fire, but increase again as the understorey vegetation regenerates (Friend *et al.* 1999). There is also a longer-term seral response by mammals, with species increasing when conditions become suitable for them, then decreasing again as the habitat changes and favours other species (Fox and McKay 1981; Friend 1993). Seral responses differ depending on the shelter and resource requirements of particular species, as well as their reproductive patterns (Friend 1993), but may also vary depending on the severity of the fire. For example, burning of logs or dead trees will have a longer-term detrimental impact on species that use them, such as the Yellow-footed Antechinus (*Antechinus flavipes*), Brush-tailed Phascogale (*Phascogale tapoatafa*) and Short-beaked Echidna (*Tachyglossus aculeatus*) (Andrew Bennett, Deakin University, pers. comm.).

In dry forests and heathlands, the presence and abundance of several small ground mammals have been correlated with time since fire, with early colonisers often being the introduced Black Rat (*Rattus rattus*) (Robertson 1985) or House Mouse (*Mus musculus*) (Fox and McKay 1981; Friend 1993; Meredith 1987; Newsome *et al.* 1975; Recher *et al.* 1974; Robertson 1985). Dunnarts (*Sminthopsis* spp.) generally prefer the early stages of regeneration (Irvin *et al.* 2003b), while Bush Rats (*Rattus fuscipes*) and Brown Antechinus (*Antechinus stuartii*) may colonise for up to 10 years after fire (Meredith 1987).

In the Wombat State Forest, only the Bush Rat and Brown Antechinus existed in sufficient numbers to allow statistical analysis (Irvin *et al.* 2003b). The study found that the Brown Antechinus was dependent on leaf litter and logs for foraging habitat, and declined significantly after fire. Populations were affected more by autumn than spring burns due to the time of breeding, but they nonetheless recovered by the second year post-fire (Irvin *et al.* 2003b). Similarly, after the 1969 summer wild fire near Daylesford, the population of Brown Antechinus was reduced by 90%, but had recovered to similar levels to the unburnt vegetation after two breeding seasons (Heislers 1974). In contrast, in dry sclerophyll forest in Western Australia, populations of Yellow-footed Antechinus had not recovered 4 ½ years after fire, and the greatest numbers were found in forest that had not been burnt for many years (Christensen and Kimber 1975).

Small mammals were particularly disadvantaged by fierce wildfire in dry coastal sclerophyll forest, with the Swamp Rat (*Rattus lutreolus*), Bush Rat, Dusky Antechinus and Brown Antechinus virtually absent two years later (Newsome *et al.* 1975). Similarly, small mammals had not re-colonised heathland 15 months after being burnt (Recher *et al.* 1974). Some species may be affected by even low intensity fires. For example, the Brown Antechinus, which is dependent on a high percentage of ground cover, was not seen in the three years after fire near Bega, New South Wales (Lunney *et al.* 1987). Interestingly, Brown Antechinus did not decline in number after fire in sclerophyll forest at Leonards Hill and Barkstead near Ballarat, even when one site had little ground cover (Leonard 1972). The author suggested that this species was less dependent on cover than the Bush Rat or Dusky Antechinus.

The Bush Rat is dependent on dense, sedge-dominated vegetation in or near gullies (Braithwaite *et al.* 1978; Irvin *et al.* 2003b), and is substantially less abundant in drier forest (Bennett 1993). Following a single autumn or spring fire in the Wombat State Forest, populations declined significantly, with recovery taking one to three breeding seasons, depending on the amount of the habitat burnt (Irvin *et al.* 2003b). However, the Bush Rat generally thrives 3-5 years post-fire (Fox and McKay 1981; Irvin *et al.* 2003b), or even within two years when the cover of herbs and grasses in gullies has increased substantially (Heislers 1974). The recovery of vegetation is an important factor for mammals, as studies have found that succession after fire is related more to vegetation density than time since fire *per se* (Friend and Wayne 2003; Monamy and Fox 2000).

Despite their ability to recover, the survival of both the Brown Antechinus and Bush Rat relies on the retention of unburnt habitat patches (Irvin *et al.* 2003b), and populations are likely to decrease if fire frequency averages 8-10 years (Meredith 1987). Interestingly, the generation of Bush Rats alive at the time of fire at Leonards Hill near Ballarat did not decrease in numbers: the effect was noted in the following generation (Leonard 1972). This suggests that timing of fauna surveys might affect the interpretation of fire responses.

In South Australia, the Southern Brown Bandicoot (*Isoodon obesulus*) prefers mid-successional habitat, and management plans for conservation of this animal prescribe mosaic burning at 10 to 14 year rotations (Possingham and Gepp 1996). The Brown Antechinus and Dusky Antechinus typically occupy late successional stages in heathland (Friend 1993), due to their relatively specific and inflexible life history characteristics. In general, most small mammal species in dry forests are unlikely to be disadvantaged by fires where the inter-fire period is at least 15-20 years (Meredith 1987). However, this may not apply to Box-Ironbark remnants, where recolonisation by small mammals after fire may be severely hampered by isolation. For example, roads in wetter forests in north-eastern New South Wales, even partly overgrown tracks, prevented or severely restricted the movement of species such as Brown Antechinus, Bush Rat and Fawn-footed Melomys (*Melomys cervinipes*) (Barnett 1978).

Few useful studies have been undertaken on the impact of fires on larger mammals (Meredith 1987), and no large animals existed in sufficient numbers in the Wombat State Forest to allow quantitative analysis (Irvin *et al.* 2003b). High mortality of macropods has been observed following intense fire (Heislers 1974; Newsome *et al.* 1975), although four years after the 1969 fire near Daylesford the population numbers of kangaroos and wallabies were at least as high as in unburnt forest (Heislers 1974). However, an ecological burn in a patch of remnant Box-Ironbark is highly unlikely to be of a size or intensity likely to cause direct mortality. Low intensity fires are more likely to draw larger animals in, as recently burnt areas are attractive to grazing and browsing animals (Christensen and Kimber 1975; Robertson 1985). For example, in dry sclerophyll forest, new shoots of *Poa* tussock-grass and shrubs are preferentially grazed following burning (Tolhurst and Oswin 1992), and transient increases in the abundance of kangaroos and wallabies may be expected. Post-fire grazing is more pronounced following autumn burning, probably as a consequence of other green fodder being limited. Similarly, increases in the abundance of the Brush Wallaby (*Macropus irma*) and Western Grey Kangaroo (*Macropus fuliginosus*) were noted in dry sclerophyll forest in Western Australia in the first two or three years after fire (Christensen and Kimber 1975).

Large burrowing mammals such as wombats are capable of surviving even high-intensity fire (Friend *et al.* 1999; Newsome *et al.* 1975), and may also be able to take advantage of nutritious regeneration in the post-fire period. For some animals, predation following fire may be significant, and studies in Western Australia have found that in some instances the effects of this predation on various species have been greater than the effects of the fire itself (Friend *et al.* 1999). This suggests that predator control might be an important aspect of fire planning if a burn is going to be large.

Little is also known on the effects of fire on arboreal mammals such as possums and bats (Meredith 1987; Wilson 1996; Wilson and Friend 1999), and there may be little impact if canopy trees are not burnt. Surveys after the Daylesford fire yielded low numbers of possums even after three years, and with relatively low rates of reproduction species such as the Common Ringtail Possum (*Pseudocheirus peregrinus*) may take years to recover (Heislers 1974). In Western Australia, the abundance of the Western Ringtail Possum (*Pseudocheirus occidentalis*) was greatest in sites that had not been burnt for at least 20 years, or had only been burnt with very low fire intensity (Wayne *et al.* 2005).

In Jarrah and Marri forests in Western Australia, den trees used by the Common Brushtail Possums (*Trichosurus vulpecula*) and Common Ringtail Possums were mostly the mature, over-mature or dead trees in a stand, and damage to these trees during high intensity fire (and subsequent loss of the useable hollows) tended to be proportionately greater than for younger or healthier trees (Inions *et al.* 1989). Nonetheless, the possums were able to adjust to the changes in condition, utilising new or enlarged hollows in younger trees (Inions *et al.* 1989). However, mammals such as Ringtail Possums may also nest in shrubby vegetation dominated by species such as *Melaleuca* or *Acacia*, and may be severely impacted by even low-intensity fires in the short term (Meredith 1987; Newsome *et al.* 1975). However, Box-ironbark forests usually lack such shrub thickets, and Common Ringtail Possums are essentially hollow-dependent (Soderquist and Mac Nally 2000). Occasional fire may accelerate the development of tree hollows in the long term (Meredith 1987), by producing scars that are susceptible to fungal or termite attack (Adkins *et al.* 2005; Inions *et al.* 1989).

Position in the landscape, as noted for birds, is also a major factor in the abundance of mammals. Population densities of the Common Brushtail Possum and Common Ringtail Possum were around three times higher in gullies than on ridges (Soderquist and Mac Nally 2000), and nearly all reproduction occurred in gullies. The Yellow-footed Antechinus was also more abundant in gullies (Soderquist and Mac Nally 2000).

Bats occur in large numbers in many forests, yet are sometimes overlooked in mammal surveys (Irvin *et al.* 2003b). Seven species occur in the study sites in the Wombat State Forest, making bats the most species-rich mammal group. There was no significant difference in bat activity between burn treatments (Irvin *et al.* 2003b), but the large home range of these animals relative to the size of the study sites made it difficult to draw conclusions.

The impact of fire on mammals may be obscure, involving complex relationships with plants and other lifeforms. For example, hypogeous fungi may form a large portion of the diet of Black Wallabies (*Wallabia bicolor*) (Claridge *et al.* 2001; Hollis *et al.* 1986) and the wallabies may play an important role in maintaining fungal diversity by dispersing spores (Claridge *et al.* 2001). Further, faeces of mycophagous marsupials are then considered to be reliable sources of ectomycorrhizal spores which are important in the regeneration of mycorrhiza-forming plants, especially after fire

(Claridge *et al.* 2001; Lamont *et al.* 1985). There is some evidence from Western Australia that long fire-free periods may lead to a reduction in hypogeal fungi (Lamont *et al.* 1985), with potential flow-on effects on mycorrhiza-forming plants. However, this aspect has not been studied in Box-Ironbark forests.

Overall, there appears to be no single fire regime that will encourage maximum populations of all the mammal species in a forest ecosystem (Christensen and Kimber 1975; Suckling and Macfarlane 1984; Wilson 1996). Frequent fire will advantage some small mammals, while infrequent fire may advantage others. Spring fires may have a greater impact than autumn fires, by disrupting the breeding season of small mammals such as dasyurids (Wilson 1996). In privately-owned, often small and mostly disturbed Box-Ironbark remnants, the abundance and composition of native mammals are likely to be substantially altered in comparison to those resident in larger tracts of relatively undisturbed forest. Burning of small remnants in their entirety is likely to be of further detriment, as the fragmented nature of most remnants may act as a barrier to recolonisation. However, partial burning of those small remnants may not provide sufficient stimulus for biodiversity improvement.

Fifty-four mammal species have been recorded across the broader Box-Ironbark study area (Silveira *et al.* 1997). For the purpose of this report, species that are locally extinct or marginal have been omitted, leaving 26 native and 7 introduced mammals that are likely to be in the forests of interest. Expert opinion was sought to gauge the impact (positive or negative, long-term or short-term) on these mammals by applied fire of low-moderate intensity. Species that are expected to respond strongly are shown in Table 4, while the full species list is provided in Appendix 4.

Table 4. Mammal species likely to be most affected by a low-moderate intensity fire in Box-Ironbark forest. HP = highly positive, SP = slightly positive. * = introduced.

Common name	Scientific name	Spring burn, short-term	Spring burn, long-term	Autumn burn, short-term	Autumn burn, long-term
Western Grey Kangaroo	*Macropus fuliginosus*	**HP**	**HP**	SP	SP
Eastern Grey Kangaroo	*Macropus giganteus*	**HP**	**HP**	SP	SP
Black Wallaby	*Wallabia bicolor*	**HP**	**HP**	SP	SP
Brown Hare*	*Lepus capensis*	**HP**	**HP**	SP	SP
European Rabbit*	*Oryctolagus cuniculus*	**HP**	**HP**	SP	SP

Only five mammal species (the three major macropods, and introduced rabbits and hares) are expected to show a strong positive response to fire in Box-Ironbark forest (Table 4). These are all common herbivorous species that should benefit from the flush of new herb or grass growth encouraged by spring burns. Autumn burns are likely to elicit only a slight positive response by these species, commensurate with a smaller plant response. Some introduced mammals, especially rodents, are likely to show a slightly positive response, being highly fecund generalists (Appendix 4). No species are expected to be highly negatively affected by either spring or autumn low-intensity burns, although many small mammals may be slightly negatively affected due to their dependence on the ground layer. The impact on possums, gliders and bats is likely to be neutral, unless fire severity is high.

Key findings

- The effects of fire on mammals are closely linked to the effects and recovery of the vegetation, and the size and intensity of the burn.
- Some common herbivores and rodents will be able to take advantage of the flush of new plant growth, but small animals dependent on the ground layer will be disadvantaged in the short-term.
- Arboreal mammals are likely to be negatively affected only if the fire is of high intensity.
- In general, it is believed that small mammals will not be disadvantaged provided the minimum inter-fire period is at least 15-20 years.
- In isolated private remnants, recolonisation by small mammals after fire may be difficult.
- Burning of habitat such as logs should be avoided.
- Burning of gully vegetation should be avoided.
- Further research is required, particularly in regard to the response of arboreal mammals such as possums and bats.

Effects of Fire on Reptiles and Frogs

There are few data on the effects of fire on reptiles, particularly in southern Australia (Friend *et al.* 1999; Irvin *et al.* 2003a; Meredith 1987; Wilson 1996), and most studies have concentrated on areas of relatively high reptile diversity.

The lower metabolic needs of reptiles associated with ectothermy mean that these animals are less susceptible than endothermic animals (mammals and birds) to short-term food shortages that may be associated with fire. Nonetheless, being directly dependent on the ground layer for food and shelter (Bennett 1999), burning has consequences in terms of both direct mortality and post-fire persistence. For example, in south-east Queensland open sclerophyll forest, the abundance of the Lively Rainbow Skink (*Carlia vivax*) was reduced in burnt areas compared to unburnt areas, due to its preference for ground and litter cover (Singh *et al.* 2002). Thus, even low-intensity burning can have implications for individual reptile species. In the first year after burning, predation and intraspecific competition are also believed to be major factors in reducing lizard abundance (Irvin *et al.* 2003a). Reptile survival is therefore dependent on there being a small-scale mosaic of burnt and unburnt patches, with fallen logs and unburnt litter providing refuge (Irvin *et al.* 2003a).

In mixed eucalypt foothill forest (Wombat State Forest, central Victoria), Coventry's Skink (*Pseudemoia coventryi*) and Grass Skink (*P. entrecasteauxii*) forage in the grass/herb and litter layers, which are most affected by fire (Irvin *et al.* 2003a). The numbers of both species were significantly reduced in the years after fire, particularly when frequently burnt in either autumn or spring (Scuffins 1994, cited in Irvin *et al.* 2003a). In contrast, the abundance of the Southern Water Skink (*Eulamprus tympanum*), a lizard that relies on logs and fallen branches, remained relatively stable (Scuffins 1994, cited in Irvin *et al.* 2003a). The Garden Skink (*Lampropholis guichenoti*) was only affected if a significant amount of vegetation was burnt (Irvin *et al.* 2003a), while McCoys Skink (*Nannoscincus maccoyi*) was more evident following burning because of its normally cryptic nature (Humphries 1994, cited in Irvin *et al.* 2003a). However, this species is dependent on deep litter and could be eliminated by high-frequency fire (Irvin *et al.* 2003a). In general, all species observed before burning were also found after burning, although the abundances of individual species differed (Irvin *et al.* 2003a).

Studies in mallee vegetation have shown that the number of reptile species does not differ over time as vegetation ages after fire, but relative abundances of particular species do (Caughley 1985; Schlesinger *et al.* 1997). The species that were common soon after fire preferred open ground for foraging (Caughley 1985; Schlesinger *et al.* 1997), and were all uncommon in old growth vegetation (Caughley 1985). In contrast, five other species were more common in 25-year-old regrowth. This reflects the range of ecological requirements and habitats required by these animals, and suggests that changes in vegetation over time will alter its suitability for individual species (as in the seral response of small mammals). Differences in species response were also noted 20 months after fire in Ash/Stringybark forest near Bega (New South Wales), where numbers of Delicate Skink (*Lampropholis delicata*) remained depressed, Garden Skink had initially declined but recovered, and Yellow-bellied Water Skink (*Eulamprus heatwolei*) was relatively unaffected (Penn *et al.* 2003). The effects of fire on the Garden Skink were found to be more severe on ridges than in gullies, because of higher fire intensity and subsequent scarcity of cover and food (Lunney *et al.* 1991). However, just as fire is not a uniform disturbance, species response may also be inconsistent. For example, the research by Lunney *et al.* (1991) and Penn *et al.* (2003) was undertaken after different fires in the same forest, and came to slightly different conclusions. For example, numbers of Delicate Skink were higher, rather than lower, after the earlier fire.

Predicting the response of reptiles to fire in Box-Ironbark forest is difficult, as the reasons for the response patterns observed after fire are not always clear. Much of the research involves multiple agents of disturbance, with drought in particular being a confounding factor. The richness and abundance of reptiles in Box-Ironbark forest is typically low (Brown 2001), and lower still in disturbed Box-Ironbark remnants, which may have only around 40% of the number of individuals found in less-disturbed sites, and significantly fewer species (Bennett 1999; Brown 2001). Nestedness was found in Box-Ironbark fragments with respect to reptile fauna (Mac Nally and Brown 2001), thus reptiles in smaller fragments may be more vulnerable, especially if fire is applied to most or all of the fragment. Given the already low species richness, the impact of fire on reptiles in small remnant vegetation patches might be to drive it lower still. However, no relevant research was found.

In some forest types, an occasional fire might be useful in reducing shrub and ground cover, hence increasing available basking sites (Irvin *et al.* 2003a). However, Box-Ironbark forests when not highly modified are relatively open with discontinuous ground cover, and it is unlikely that such structural opening is needed. Fire is more likely to add some structural complexity by promoting leguminous shrub species, and this added complexity might favour some species. However, no research was found that addresses this. Further, Box-Ironbark woodlands are distinguished by a relatively high degree of fragmentation and isolation (Brown 2001). Therefore, even if the habitat quality of many sites could be enhanced by the occasional applied fire, thereby improving the survival probability of remaining animals, the geographical isolation of private remnants may be a barrier to recolonisation for some species.

There are few data on the effects of fire on frogs (Bamford 1992; Driscoll and Roberts 1997; Friend *et al.* 1999; Meredith 1987; Wilson 1996), and the limited available research, much of it from Western Australia, shows inconsistent post-fire trends. Some frog species burrow and have generalized invertebrate diets, and thus show little relationship with fire (Friend *et al.* 1999). Fire and time-since-fire did not appear to have an impact on the frog fauna of woodlands near Perth, Western Australia (Bamford 1992). In contrast, spring fuel-reduction burns in forest south of Perth had a significant impact on the Walpole Frog (*Geocrinia lutea*), with a significant decline in the number of calling males, and no recovery two years later (Driscoll and Roberts 1997).

By utilising non-flammable refugia such as wet areas (Russell *et al.* 1999) and subterranean shelter, many frogs may be largely protected from the direct impacts of fire, although nomadic species may disperse away from the burnt areas after such fire (Wilson 1996). However, some species in Box-Ironbark do not burrow (Barker *et al.* 1995), and may be directly exposed to fire. Increases in run-off after fire may have longer-term implications for water quality and frog habitat (Brown *et al.* 1998), particularly in areas that experience repeated or intense fires.

In some Australian ecosystems, the majority of frog species tend to breed in the cooler and wetter months, and their distribution and abundance may be more closely tied to moisture than vegetation succession (Friend 1993). However, many of the frogs that occur in Box-Ironbark forest will breed in the warmer months if conditions are suitable (Hero *et al.* 1991; Tzaros 2005), and their reproductive cycle might be affected by fire. However, no research was found that addressed this issue in Box-Ironbark or similar ecosystems.

Previous surveys in Box-Ironbark forest (Arthur Rylah Institute, unpublished data) have identified 43 reptile and 14 frog species that are likely to be in the study area. Expert opinion was sought to gauge the impact (positive or negative, long-term or short-term) on these Box-Ironbark herpetofauna species by applied fire of low-moderate intensity. Species that are expected to respond strongly are shown in Table 5, while the full species list is provided in Appendix 5.

Only five lizard species and two frog species are expected to show a strong negative response to fire in Box-Ironbark forest (Table 5). The lizards are all ground-dwelling/fossorial species that breed, and are most active, in spring, and are likely to be highly negatively impacted in the short-term by a fire at that time. Their eggs may also be at risk from fire. The impact of spring fire in the long-term is expected to be slightly negative, as is the short-term effect of autumn burning.

The frogs are both threatened species suspected to be on the decline. They are ground-dwelling animals that lay egg masses in soil or litter in autumn, and would be severely disadvantaged by a fire at that time (Table 5). The impact of a spring burn is likely to be slightly negative.

The impact of low-intensity spring or autumn fire on other reptiles is likely to be slightly negative in the short-term (Appendix 5), due to their dependence on the ground layer. In the long-term, the effects are mostly neutral or unknown. The impact of a spring burn on other frog species is likely to be slightly negative in the short-term, but the impact of autumn burning is largely unknown (Appendix 5).

Table 5. Reptile and frog species likely to be most affected by a low-moderate intensity fire in Box-Ironbark forest. HN = highly negative, SN = slightly negative, U = unknown.

Common name	Scientific name	Spring burn, short-term	Spring burn, long-term	Autumn burn, short-term	Autumn burn, long-term
Striped Legless Lizard	*Delma impar*	**HN**	SN	SN	U
Olive Legless Lizard	*Delma inornata*	**HN**	SN	SN	U
Burton's Snake-Lizard	*Lialis burtonis*	**HN**	SN	SN	U
Common Scaly-foot	*Pygopus lepidopodus*	**HN**	SN	SN	U
Hooded Scaly-foot	*Pygopus schraderi*	**HN**	SN	SN	U
Bibron's Toadlet	*Pseudophryne bibronii*	SN	U	**HN**	SN
Southern Toadlet	*Pseudophryne semimarmorata*	SN	U	**HN**	SN

Key findings

- Many reptiles and amphibians are likely to be negatively affected in the short-term by any burning due to their dependence on the litter and ground layers.
- Spring burning will affect the breeding of many reptile species, while autumn burning may affect the breeding of some frog species.
- In remnants, isolation is likely to be a barrier to recolonisation after fire.
- Further research is required on this fauna group.

Summary of Effects of Fire on Fauna

The effects of fire on the various fauna groups will differ depending on the season of burn, fire intensity, size of burn, and breeding and habitat requirements. Few generalisations are possible, but the available literature suggests that most fauna communities are unlikely to be adversely affected in the long-term provided fire is not too frequent. Broad responses by the fauna groups to low intensity fire are summarised in Table 6.

Table 6. Summary of fauna responses to low intensity fire in Box-Ironbark forests.

Fauna Group	Short-term effect of fire	Long-term effect of fire	Approximate minimum inter-fire period
Invertebrates	Insects reliant on the litter layer, such as Springtails, earthworms and some beetles, will decline. Ants may increase.	Little effect, unless burning is frequent. Effects on termites unknown.	3 years.
Birds	Ground-feeding and nesting species adversely affected, especially by spring burn. Raptors advantaged by increased prey availability.	Seed-eaters advantaged by increased seed production. Some ground-nesters or species reliant on litter layer slightly disadvantaged, but effects generally neutral or slightly positive.	Linked to recovery of understorey: around 25 years.
Mammals	Macropods and introduced herbivores and rodents advantaged. Small mammals reliant on ground layers disadvantaged.	Small mammals reliant on ground layers slightly disadvantaged. Effects on other mammals generally neutral or slightly positive.	15-20 years.
Reptiles and Frogs	Ground-dwelling/fossorial species will be disadvantaged, especially by spring burns.	Slightly negative on some ground-dwelling species, but mostly neutral or unknown.	Not known.

CONCLUSION

The effects of fire on flora and fauna in Box-Ironbark forests will vary depending on factors such as fire intensity, size of fire, season of burn, site factors, landscape position, soil type, seed bank, disturbance history, edge effects and the life characteristics of individual flora and fauna species.

Few plant species in Box-Ironbark forests rely on fire for germination, and most species will continue to recruit, albeit at a reduced level. No plant species should be lost under an appropriate fire regime, provided the inter-fire period allows all species to reach reproductive maturity (a minimum of 10-20 years). Most species will persist even when the interval between fires exceeds 50 years. Nonetheless, occasional fire may be useful for ensuring that the soil seed bank for individual species is not eventually depleted. However, fire is unlikely to restore 'lost' species in small or disturbed remnants, and a burning regime in these remnants may only maintain rather than enhance diversity. No single burning regime will advantage or disadvantage every species group, and an appropriate burning regime will ensure a mosaic of different age classes. The vegetation structure and diversity of Box-Ironbark remnants will vary depending on disturbance history, and each site will need to be assessed individually. Criteria for determining the most appropriate fire regime will be elaborated in Phase 2 of the project.

The effects of fire on fauna are usually linked to the effects on and recovery of the vegetation structure. Effects are mostly short-term, and few faunal communities will be adversely affected provided the inter-fire period allows full recovery of that vegetation structure (around 25 years). However, the abundance and species composition in small or disturbed remnants may be substantially different to those in larger or relatively undisturbed blocks, making the effects of fire on remnants difficult to predict. Isolation of remnants may also prove to be a barrier to recolonisation for some species after fire, especially reptiles, frogs and small mammals. There is insufficient evidence to justify or formulate an ecological burning regime for the maintenance or enhancement of faunal diversity in small Box-Ironbark remnants. However, a regime appropriate to the vegetation, which provides for a mosaic of age classes, is unlikely to adversely affect fauna groups in larger blocks.

REFERENCES

Adams R. and Simmons D. (1996) The impact of fire intensity on litter loads and understorey floristics in an urban fringe dry sclerophyll forest and implications for management. In '*Fire and Biodiversity: The Effects and Effectiveness of Fire Management*'. 8-9 October 1994, Footscray, Melbourne. (Department of the Environment, Sport and Territories)

Adkins M.F., Westbrooke M.E., Florentine S.K. and McDonald S.P. (2005) Fire and hollow formation in box-ironbark eucalypts of the Warby Range State Park. *The Victorian Naturalist* **122**, 47-56.

Andersen A.N. and Yen A.L. (1985) Immediate effects of fire on ants in the semi-arid mallee region of north-western Victoria. *Australian Journal of Ecology* **10**, 25-30.

Andrew N., Rodgerson L. and York A. (2000) Frequent fuel-reduction burning: the role of logs and associated leaf litter in the conservation of ant biodiversity. *Austral Ecology* **25**, 99-107.

Auld T.D. and Tozer M. (1995) Patterns in emergence of Acacia and Grevillea seedlings after fire. *Proceedings of the Linnean Society of New South Wales* **115**, 5-15.

Backhouse G.N. (1987) Management of remnant habitat for conservation of the Helmeted Honeyeater Lichenostomus melanops cassidix. In *Nature Conservation: The Role of Remnants of Native Vegetation.* (Eds D.A. Saunders, G.W. Arnold, A. Burbidge and A.J.M. Hopkins) pp. 287-294. (Surrey Beatty, Chipping Norton)

Bamford M.J. (1992) The impact of fire and increasing time after fire upon Heleioporus eyrei, Lymnodynastes dorsalis and Myobatrachus gouldii (Anura: Leptodactylidae) in Banksia woodland near Perth, Western Australia. *Wildlife Research* **19**, 169-178.

Barker J., Grigg G.C. and Tyler M.J. (1995) *A Field Guide to Australian Frogs.* (Surrey Beatty & Sons, Chipping Norton)

Barnett J.L. (1978) The use of habitat components by small mammals in eastern Australia. *Australian Journal of Ecology* **3**, 277-285.

Bell D.T. (1999) Turner Review No. 1. The process of germination in Australian species. *Australian Journal of Botany* **47**, 475-517.

Bennett A. (1999) Wildlife of the Box-Ironbark forests: understanding pattern and process. *Trees and Natural Resources*, 26-28.

Bennett A., Mac Nally R. and Yen A.L. (1999) *Extinction Processes and Fauna Conservation in Remnant Box-Ironbark Woodlands.* Final report to Land and Water Resources R & D Corporation and Environment Australia.

Bennett A.F. (1993) Microhabitat use by the Long-nosed Potoroo, Potorous tridactylus, and other small mammals in remnant forest vegetation of south-western Victoria. *Wildlife Research* **20**, 267-285.

Benwell A.S. (1998) Post-fire seedling recruitment in coastal heathland in relation to regeneration strategy and habitat. *Australian Journal of Botany* **46**, 75-101.

Bowman D.M.J.S. (1998) The impact of Aboriginal landscape burning on the Australian biota. *New Phytologist* **140**, 385-410.

Braithwaite R.W., Cockburn A. and Lee A.K. (1978) Resource partitioning by small mammals in lowland heath communities of south-eastern Australia. *Australian Journal of Ecology* **3**, 423-445.

Brooker M.G. (1998) Fire and birds in a Western Australian heathland. *Emu* **98**, 276-287.

Brown G.W. (2001) The influence of habitat disturbance on reptiles in a box-ironbark eucalypt forest of south-eastern Australia. *Biodiversity and Conservation* **10**, 161-176.

Brown J., Enright N.J. and Miller B.P. (2003) Seed production and germination in two rare and three common co-occurring Acacia species from south-east Australia. *Austral Ecology* **28**, 271-280.

Brown K.L., Gadd L.S., Norton T.W., Williams J.E. and Klom N.I. (1998) *The Effects of Fire on Fauna in the Australian Alps National Parks: A Database*. Report No 118. The Johnstone Centre, Charles Sturt University.

Calder M., Calder J. and McCann I. (1994) *The Forgotten Forests: A Guide to Victoria's Box and Ironbark Country*. (Victorian National Parks Association, Melbourne)

Campbell A.J. and Tanton M.T. (1981) Effects of fire on the invertebrate fauna of soil and litter of a eucalypt forest. In *Fire and the Australian Biota*. (Eds A.M. Gill, R.H. Groves and I.R. Noble) pp. 215-241. (Australian Academy of Science, Canberra)

Catling P.C. and Burt R.J. (1995) Studies of the ground-dwelling mammals of eucalypt forests in south-eastern New South Wales: the effect of habitat variables on distribution and abundance. *Wildlife Research* **22**, 271-288.

Caughley J. (1985) Effect of fire on the reptile fauna of mallee. In *Biology of Australian Frogs and Reptiles*. (Eds G. Grigg, R. Shine and H. Erman). (Surrey Beatty, Chipping Norton)

Chatto K. (1996) *Fuel Hazard Levels in Relation to Site Characteristics and Fire History - Chiltern Regional Park Study Case. Research Report No. 43*. Department of Natural Resources and Environment, Victoria.

Cheney P. (1996) The effectiveness of fuel reduction burning for fire management. In '*Fire and Biodiversity: The Effects and Effectiveness of Fire Management*'. 8-9

October 1994, Footscray, Melbourne. (Department of the Environment, Sport and Territories)

Chesterfield E.A. (1984) Effects of fire on the flora of dry sclerophyll forests. In '*Fighting Fire With Fire*'. Proceedings of symposium on fuel reduction burning held at Monash University, September 17-18, 1983. (Ed. E.H.M. Ealey)

Christensen P., Recher H. and Hoare J. (1981) Responses of open forests (dry sclerophyll forests) to fire regimes. In *Fire and the Australian Biota*. (Eds A.M. Gill, R.H. Groves and I.R. Noble) pp. 367-393. (Australian Academy of Science, Canberra)

Christensen P.E. and Kimber P.C. (1975) Effect of prescribed burning on the flora and fauna of south-west Australian forests. *Proceedings of the Ecological Society of Australia* **9**, 85-106.

Claridge A.W., Trappe J.M. and Claridge D.L. (2001) Mycophagy by the swamp wallaby (Wallabia bicolor). *Wildlife Research* **28**, 643-645.

Clarke P.J. and Knox K.J.E. (2002) Post-fire response of shrubs in the tablelands of eastern Australia: do existing models explain habitat differences? *Australian Journal of Botany* **50**, 53-62.

Clarke P.J., Knox K.J.E., Wills K.E. and Campbell M. (2005) Landscape patterns of woody plant response to crown fire: disturbance and productivity influence sprouting ability. *Journal of Ecology* **93**, 544-555.

Collett N. (2003) Short and long-term effects of prescribed fires in autumn and spring on surface-active arthropods in dry sclerophyll eucalypt forests of Victoria. *Forest Ecology and Management* **182**, 117-138.

Collett N. and Neumann F. (2003) *Effects of Repeated Low-Intensity Fire on the Invertebrates of a Mixed Eucalypt Foothill Forest in South-Eastern Australia. Research Report No. 61*. Department of Sustainability and Environment, Melbourne.

Collett N.G. (1998) Effects of two short rotation prescribed fires in autumn on surface-active arthropods in dry scerophyll eucalypt forest of west-central Victoria. *Forest Ecology and Management* **107**, 253-273.

Collett N.G. (1999) Effects of three short rotation prescribed fires in spring on surface-active arthropods in dry sclerophyll eucalypt forest of west-central Victoria. *Australian Forestry* **62**, 295-306.

Collett N.G. and Neumann F.G. (1995) Effects of two spring prescribed fires on epigeal Coleoptera in dry sclerophyll eucalypt forest in Victoria, Australia. *Forest Ecology and Management* **76**, 69-85.

Collett N.G., Neumann F.G. and Tolhurst K.G. (1993) Effects of two short rotation prescribed fires in spring on surface-active arthropods and earthworms in dry sclerophyll eucalypt forest of west-central Victoria. *Australian Forestry* **56**, 49-60.

DCNR (1995) *Code of Practice for Fire Management on Public land.* Department of Conservation and Natural Resources, East Melbourne.

Dettmann P.D., Hamilton S.D. and Curtis A.L. (2000) Understanding landholder values and intentions to improve remnant vegetation management in Australia: the box-ironbark case study. *Journal of Sustainable Agriculture* **16**, 93-105.

Driscoll D.A. and Roberts J.D. (1997) Impact of fuel-reduction burning on the frog Geocrinia lutea in southwest Western Australia. *Australian Journal of Ecology* **22**, 334-339.

DSE (2003a) *Advisory List of Threatened Vertebrate Fauna in Victoria - 2003.* Department of Sustainability and Environment, East Melbourne, Victoria.

DSE (2003b) *North West Region: Bendigo Fire District Fire protection Plan.* Department of Sustainability and Environment, East Melbourne.

DSE (2006) *Draft Ecological Fire Strategy for Northern Bendigo Landscape Management Unit.* Department of Sustainability and Environment & Parks Victoria, Bendigo, Victoria.

ECC (1997) *Box-Ironbark Forests and Woodlands Investigation Resources and Issues Report.* Environment Conservation Council, Fitzroy.

ECC (2000) *Box-Ironbark Forests and Woodlands Investigation: Draft for Public Comment.* Environment Conservation Council, East Melbourne.

ECC (2001) *Box-Ironbark Forests and Woodlands Investigation: Final Report.* Environment Conservation Council, East Melbourne.

Fox B.J. and McKay G.M. (1981) Small mammal responses to pyric successional changes in eucalypt forest. *Australian Journal of Ecology* **6**, 29-41.

Friend G. (1996) Fire ecology of invertebrates - implications for nature conservation, fire management and future research. In '*Fire and Biodiversity. The Effects and Effectiveness of Fire Management*'. Proceedings of the Conference held 8-9 October 1994, Footscray, Melbourne. (Department of the Environment Sport and Territories, Canberra)

Friend G., Leonard M., MacLean A. and Sieler I. (1999) (Eds) *Management of Fire for the Conservation of Biodiversity - Workshop Proceedings.* (Department of Natural Resources and Environment, Melbourne)

Friend G. and Wayne A. (2003) Relationships between mammals and fire in south-western Australian ecosystems: what we know and what we need to know. In *Fire in South-Western Australian Ecosystems: Impacts and Management.* (Eds I. Abbott and N.D. Burrows) pp. 363-380. (Backhuys Publishers, Leiden)

Friend G.R. (1993) Impact of fire on small vertebrates in mallee woodlands and heathlands of temperate Australia: a review. *Biological Conservation* **65**, 99-114.

Friend G.R. (1995) Fire and invertebrates - a review of research methodology and the predictability of post-fire response patterns. *CALM Science Supplement* **4**, 165-174.

Gill A.M. (1981) Adaptive responses of Australian vascular plant species to fires. In *Fire and the Australian Biota.* (Eds A.M. Gill, R.H. Groves and I.R. Noble) pp. 243-272. (Australian Academy of Science, Canberra)

Good R.B. (1981) The role of fire in conservation reserves. In *Fire and the Australian Biota*. (Eds A.M. Gill, R.H. Groves and I.R. Noble) pp. 529-549. (Australian Academy of Science, Canberra)

Gott B. (2005) Aboriginal fire management in south-eastern Australia: aims and frequency. *Journal of Biogeography* **32**, 1203-1208.

Green K. and Sanecki G. (2006) Immediate and short-term responses of bird and mammal assemblages to a subalpine wildfire in the Snowy Mountains, Australia. *Austral Ecology* **31**, 673-681.

h-skis V. (2000) Vegetation changes since European settlement of Australia: an attempt to clear up some burning issues. *Australian Forestry* **63**, 166-173.

Hamilton S.D., Lawrie A.C., Hopmans P. and Leonard B.V. (1991) Effects of fuel-reduction burning on a Eucalyptus obliqua forest ecosystem in Victoria. *Australian Journal of Botany* **39**, 203-217.

Hartskeerl K. (1997) *Effects of Different Fire Regimes on the Floristics and Structure of Vegetation in Box-Ironbark Forests in the Bendigo District.*

Heislers A. (1974) Effects of wildfire on mammals in a Victorian eucalypt forest. In '*Proceedings - 3rd Fire Ecology Symposium, Monash University, Melbourne*'

Henderson M.K. and Keith D.A. (2002) Correlation of burning and grazing indicators with composition of woody understorey flora of dells in a temperate eucalypt forest. *Austral Ecology* **27**, 121-131.

Hero J.-M., Littlejohn M. and Marantelli G. (1991) *Frogwatch Field Guide to Victorian Frogs.* (Department of Conservation and Environment, East Melbourne)

Hewish M. (1983) The effect of a wildfire on birdlife in a eucalypt forest: a preliminary report on the Lerderderg Gorge seven weeks after the Wombat State Forest fire. *Geelong Naturalist* **20**, 3-15.

Hill S.J. and French K. (2004) Potential impacts of fire and grazing in an endangered ecological community: plant composition and shrub and eucalypt regeneration in Cumberland Plain Woodland. *Australian Journal of Botany* **52**, 23-29.

Hollis C.J., Robertshaw J.D. and Harden R.H. (1986) Ecology of the swamp wallaby (Wallabia bicolor) in north-eastern New South Wales. *Australian Wildlife Research* **13**.

Hopmans P. (2003) Soils: carbon, nitrogen and phosphorus. In *Ecological Effects of Repeated Low-intensity Fire in a Mixed Eucalypt Foothill Forest in South-eastern Australia - Summary Report (1984-1999). Fire Research Report No. 57.* (Department of Sustainability and Environment, Victoria)

Horton D. (2002) Fire and human societies. In '*Fire in Ecosystems of South-west Western Australia: Impacts and management*'Ascot, W.A. (Department of Conservation and Land Management)

Inions G.B., Tanton M.T. and Davey S.M. (1989) Effect of fire on the availability of hollows in trees used by the Common Brushtail Possum, Trichosurus vulpecula Kerr, 1792, and the Ringtail Possum, Pseudocheirus peregrinus Boddaerts, 1785. *Australian Wildlife Research* **16**, 449-458.

Irvin M., Westbrooke M. and Gibson M. (2003a) *Effects of Repeated Low-Intensity Fire on Reptile Populations of a Mixed Eucalypt Foothill Forest in South-Eastern Australia. Research Report No. 65.* Department of Sustainability and Environment, Melbourne.

Irvin M., Westbrooke M. and Gibson M. (2003b) *Effects of Repeated Low-Intensity Fire on Terrestrial Mammal Populations of a Mixed Eucalypt Foothill Forest in South-Eastern Australia. Research Report No. 63.* Department of Sustainability and Environment, Melbourne.

Kellas J.D. (1991) Management of the dry sclerophyll forests in Victoria. 2. Box-Ironbark forests. In *Forest Management in Australia.* (Eds F.H. McKinnell, E.R. Hopkins and J.E.D. Fox) pp. 163-169. (Surrey Beatty & Sons, NSW)

Kemp E.M. (1981) Pre-Quaternary fire in Australia. In *Fire and the Australian Biota.* (Eds A.M. Gill, R.H. Groves and I.R. Noble) pp. 3-21. (Australian Academy of Science, Canberra)

Kirkpatrick J.B. (1997) Vascular plant-eucalypt interactions. In *Eucalypt Ecology: Individuals to Ecosystems.* (Eds J.E. Williams and J.C.Z. Woinarski) pp. 227-245. (Cambridge University Press, Cambridge)

Lamont B.B., Ralph C.S. and Christensen P.E.S. (1985) Mycophagous marsupials as dispersal agents for ectomycorrhizal fungi on Eucalyptus calophylla and Gastrolobium bilobum. *New Phytologist* **101**, 651-656.

Leigh J.H. and Holgate M.D. (1979) The response of understorey species of forests and woodlands of the southern tablelands to grazing and burning. *Australian Journal of Ecology* **4**, 25-45.

Leonard B.V. (1972) *The Effect of Fire Upon Selected Small Mammals and Leaf Litter Fauna in Sclerophyll Forest in Southern Australia.* Monash University.

Leonard B.V. (1974) The effects of burning on litter fauna in eucalypt forests. In '*Proceedings - 3rd Fire Ecology Symposium, Monash University, Melbourne*'

Loyn R.H. (1997) Effects of an extensive wildfire on birds in far eastern Victoria. *Pacific Conservation Biology* **3**, 221-234.

Loyn R.H., Cunningham R.B. and Donnelly C. (2003) *Effects of Repeated Low-Intensity Fire on Bird Populations of a Mixed Eucalypt Foothill Forest in South-Eastern Australia. Research Report No. 62*. Department of Sustainability and Environment, Melbourne.

Lunney D., Cullis B. and Eby P. (1987) Effects of logging and fire on small mammals in Mumbulla State Forest, near Bega, New South Wales. *Australian Wildlife Research* **14**, 163-181.

Lunney D., Eby P. and O'Connell M. (1991) Effects of logging, fire and drought on three species of lizards in Mumbulla State Forest on the south coast of New South Wales. *Australian Journal of Ecology* **16**, 33-46.

Lunt I.D. (1998) Two hundred years of land use and vegetation change in a remnant coastal woodland in southern Australia. *Australian Journal of Botany* **46**, 629-647.

Mac Nally R., Bennett A.F. and Horrocks G. (2000a) Forecasting the impacts of habitat fragmentation. Evaluation of species-specific predictions of the impact of habitat fragmentation on birds in the box-ironbark forests of central Victoria, Australia. *Biological Conservation* **95**, 7-9.

Mac Nally R. and Brown G.W. (2001) Reptiles and habitat fragmentation in the box-ironbark forests of central Victoria, Australia: predictions, compositional change and faunal nestedness. *Oecologia* **128**, 116-125.

Mac Nally R. and Horrocks G. (1999) Landscape-scale conservation of an endangered migrant: the Swift Parrot (Lathamus discolor) in its winter range. *Biological Conservation* **92**, 335-343.

Mac Nally R., Soderquist T.R. and Tzaros C. (2000b) The conservation value of mesic gullies in dry forest landscapes: avian assemblages in the box-ironbark ecosystem of southern Australia. *Biological Conservation* **93**, 293-302.

Majer J.D. (1985) Fire effects on invertebrate fauna of forest and woodland. In *Symposium on Fire Ecology and Management in Western Australian Ecosystems*. (Ed. J.R. Ford) pp. 103-106)

Meredith C. (1984) Fire and birds - the result of two studies and their relevance to fuel reduction burning. In '*Fighting Fire With Fire*'. Proceedings of symposium on fuel reduction burning held at Monash University, September 17-18, 1983. (Ed. E.H.M. Ealey)

Meredith C. (1987) *Fire in the Victorian Environment - A Discussion Paper*. Australian Biological Research Group.

Milberg P. and Lamont B.B. (1995) Fire enhances weed invasion of roadside vegetation in southwestern Australia. *Biological Conservation* **73**, 45-49.

Monamy V. and Fox B.J. (2000) Small mammal succession is determined by vegetation density rather than time elapsed since disturbance. *Austral Ecology* **25**, 580-587.

Muir A.M., Edwards S.A. and Dickins M.J. (1995) *Description and Conservation Status of the Vegetation of the Box-Ironbark Ecosystem in Victoria.* Department of Conservation and Natural Resources, East Melbourne.

Neumann F.G., Collett N.G. and Tolhurst K.G. (1995) Coleoptera in litter of dry sclerophyll eucalypt forest and the effects of low-intensity prescribed fire on their activity and composition in west-central Victoria. *Australian Forestry* **58**, 83-98.

Newman L.A. (1961) *The Box-Ironbark Forests of Victoria, Australia.* Forests Commission of Victoria.

Newsome A.E., McIlroy J. and Catling P. (1975) The effects of an extensive wildfire on populations of twenty ground vertebrates in south-east Australia. *Proceedings of the Ecological Society of Australia* **9**, 107-123.

Nicholson P.H. (1981) Fire and the Australian Aborigine - an enigma. In *Fire and the Australian Biota.* (Eds A.M. Gill, R.H. Groves and I.R. Noble) pp. 55-76. (Australian Academy of Science, Canberra)

Noble I.R. and Slatyer R.O. (1981) Concepts and models of succession in vascular plant communities subject to recurrent fires. In *Fire and the Australian Biota.* (Eds A.M. Gill, R.H. Groves and I.R. Noble) pp. 311-335. (Australian Academy of Science, Canberra)

NRE (1996) *Kamarooka and Whipstick State Parks Management Plan.* Department of Natural Resources and Environment, East Melbourne.

NRE (1999) *Interim Guidelines and Procedures for Ecological Burning on Public Land in Victoria.* Department of Natural Resources and Environment and Parks Victoria.

Orscheg C.K. (2006) *An Investigation of Selected Ecological Processes in Ironbark Communities of the Victorian Box-Ironbark System.* PhD thesis, The University of Melbourne.

Parr C.L. and Andersen A.N. (2006) Patch mosaic burning for biodiversity conservation: a critique of the pyrodiversity paradigm. *Conservation Biology* **20**, 1610-1619.

Pausas J.G. and Bradstock R.A. (2007) Fire persistence traits of plants along a productivity and disturbance gradient in mediterranean shrublands of south-east Australia. *Global Ecology and Biogeography* **16**, 330-340.

Penn A.M., Sherwin W.B., Lunney D. and Banks P.B. (2003) The effects of a low-intensity fire on small mammals and lizards in a logged, burnt forest. *Wildlife Research* **30**, 477-486.

Possingham H. and Gepp B. (1996) Assessment of fire regime options for the southern brown bandicoot Isoodon obesulus in South Australia using population viability analysis. In '*Fire and Biodiversity. The Effects and Effectiveness of Fire Management*'. Proceedings of the Conference held 8-9 October 1994, Footscray, Melbourne. (Department of the Environment Sport and Territories, Canberra)

Prober S.M. and Thiele K.R. (1995) Conservation of the grassy white box woodlands: relative contributions of size and disturbance to floristic composition and diversity of remnants. *Australian Journal of Botany* **43**, 349-366.

Purdie R.W. (1977a) Early stages of regeneration after burning in dry sclerophyll vegetation. I. regeneration of the understorey by vegetative means. *Australian Journal of Botany* **25**, 21-34.

Purdie R.W. (1977b) Early stages of regeneration after burning in dry sclerophyll vegetation. II. regeneration by seed germination. *Australian Journal of Botany* **25**, 35-46.

PV (1997) *Paddys Ranges State Park Management Plan*. Parks Victoria, Kew.

PV (2005) *Castlemaine Diggings National Heritage Park Draft Management Plan*. Parks Victoria, Melbourne.

PV (2006a) *Greater Bendigo National Park Draft Management Plan*. Parks Victoria, Melbourne.

PV (2006b) *Heathcote-Graytown National Park and Spring Creek Nature Conservation Reserve Draft Management Plan*. Parks Victoria, Melbourne.

PV (2007) *Chiltern - Mt Pilot National Park Draft Management Plan*. Parks Victoria, Melbourne.

Recher H.F., Lunney D. and Posamentier H. (1974) Effects of wildfire on small mammals at Nadgee Nature Reserve New South Wales. In '*Proceedings - 3rd Fire Ecology Symposium, Monash University, Melbourne*'

Robertson D.J. (1985) *Interrelationships Between Kangaroos, Fire and Vegetation Dynamics at Gellibrand Hill Park, Victoria*. The University of Melbourne.

Roche S., Dixon K.W. and Pate J.S. (1998) For everything a season: smoke-induced seed germination and seedling recruitment in a Western Australian Banksia woodland. *Australian Journal of Ecology* **23**, 111-120.

Ross K.A., Fox B.J. and Fox M.D. (2002) Changes to plant species richness in forest fragments: fragment age, disturbance and fire history may be as important as area. *Journal of Biogeography* **29**, 749-765.

Russell K.R., Van Lear D.H. and Guynn Jr. D.C. (1999) Prescribed fire effects on herpetofauna: review and management implications. *Wildlife Society Bulletin* **27**, 374-384.

Schlesinger C.A., Noble J.C. and Weir T. (1997) Fire studies in mallee (Eucalyptus spp.) communities of western New South Wales: reptile and beetle populations in sites of differing fire history. *Rangeland Journal* **19**, 190-205.

Sherwin C. (1996) *The Box and Ironbark Forests and Woodlands of Northern Victoria.* Report to the Victorian National Parks Association, Melbourne.

Silveira C.E., Yen A.L., Bennett A.F., Brown G.W., Hinkley S.D., Loyn R.H., Lumsden L.F. and Smith W. (1997) *Fauna of the Box-Ironbark Study Area.* Arthur Rylah Institute for Environmental Research, Heidelberg.

Singh G., Kershaw A.P. and Clarke R. (1981) Quaternary vegetation and fire history in Australia. In *Fire and the Australian Biota*. (Eds A.M. Gill, R.H. Groves and I.R. Noble) pp. 23-54. (Australian Academy of Science, Canberra)

Singh S., Smyth A.K. and Blomberg S.P. (2002) Effect of a control burn on lizards and their structural environment in a eucalypt open-forest. *Wildlife Research* **29**, 447-454.

Smith P. (1989) Changes in a forest bird community during a period of fire and drought near Bega, New South Wales. *Australian Journal of Ecology* **14**, 41-54.

Soderquist T.R. and Mac Nally R. (2000) The conservation value of mesic gullies in dry forest landscapes: mammal populations in the box-ironbark ecosystem of southern Australia. *Biological Conservation* **93**, 281-291.

Spencer R.-J. and Baxter G.S. (2006) Effects of fire on the structure and composition of open eucalypt forests. *Austral Ecology* **31**, 638-646.

Suckling G.C. and Macfarlane M.A. (1984) The effects of fire on fauna - a review. In '*Fighting Fire With Fire. A Symposium on Fuel Reduction Burning in Forests*'. 17th September 1983, Monash University

Tolhurst K. (1999a) The fire effects research program: Wombat State Forest. In '*Management of Fire for the Conservation of Biodiversity - Workshop Proceedings*'. (Eds G. Friend, M. Leonard, A. MacLean and I. Sieler). (Department of Natural Resources and Environment, Melbourne)

Tolhurst K. (1999b) Towards the implementation of ecologically based fire regimes in the Grampians National Park. In '*Management of Fire for the Conservation of Biodiversity - Workshop Proceedings*'. (Eds G. Friend, M. Leonard, A. MacLean and I. Sieler). (Department of Natural Resources and Environment, Melbourne)

Tolhurst K. and Kelly N. (2003) Fuel dynamics. In *Ecological Effects of Repeated Low-intensity Fire in a Mixed Eucalypt Foothill Forest in South-eastern Australia - Summary Report (1984-1999). Fire Research Report No. 57.* (Department of Sustainability and Environment, Victoria)

Tolhurst K.G. (1992) Fire effects study program, Wombat State Forest - description of sites and experimental design. In *Ecological Impacts of Fuel Reduction Burning in Dry Sclerophyll Forest: First Progress report. Research Report No. 349*. (Eds K.G.

Tolhurst and D. Flinn). (Forest Research Section, Department of Conservation and Environment, Victoria)

Tolhurst K.G. (1996) Effects of fuel reduction burning on flora in a dry sclerophyll forest. In '*Fire and Biodiversity. The Effects and Effectiveness of Fire Management*'. Proceedings of the Conference held 8-9 October 1994, Footscray, Melbourne. (Department of the Environment Sport and Territories, Canberra)

Tolhurst K.G. (2003) *Effects of Repeated Low-Intensity Fire on the Understorey of a Mixed Eucalypt Foothill Forest in South-Eastern Australia. Research Report No. 58.* Department of Sustainability and Environment, Melbourne.

Tolhurst K.G. and Cheney N.P. (1999) *Synopsis of the Knowledge Used in Prescribed Burning in Victoria.* Department of Natural Resources and Environment, East Melbourne.

Tolhurst K.G. and Friend G.R. (2003) An objective basis for ecological fire management in Victoria. In '*Variability in fire interval. How important is it, and how do we manage for it?*' Proceedings of a workshop held 26 September 2003, Albany Western Australia. (Eds T.V. Walshe and J.E. Williams)

Tolhurst K.G. and Oswin D.A. (1992) Effects of spring and autumn low intensity fire on understorey vegetation in open eucalypt forest in west-central Victoria. In *Ecological Impacts of Fuel Reduction Burning in Dry Sclerophyll Forest: First Progress report. Research Report No. 349.* (Eds K.G. Tolhurst and D. Flinn). (Forest Research Section, Department of Conservation and Environment, Victoria)

Tolsma A.D. (2002) *The Effects of Fire and Grazing on the Energy Reserves of Resprouting Plants in Victoria's Alpine Grasslands.* The University of Melbourne.

Tzaros C. (2005) *Wildlife of the Box-Ironbark Country.* (CSIRO Publishing, Collingwood)

Van Heurck P. and Abbott I. (2003) Fire and terrestrial invertebrates in south-west Western Australia. In *Fire in Ecosystems of South-west Western Australia: Impacts and Management.* (Eds I. Abbott and N. Burrows) pp. 291-319. (Backhuys Publishers, Leiden)

Vanderwoude C., Andersen A.N. and House A.P.N. (1997) Community organisation, biogeography and seasonality of ants in an open forest of south-eastern Queensland. *Australian Journal of Zoology* **45**, 523-537.

Wakefield N.A. (1970) Bushfire frequency and vegetational change in south-eastern Australian forests. *The Victorian Naturalist* **87**, 152-158.

Wayne A.F., Cowling A., Lindenmayer D.B., Ward C.G., Vellios C.V., Donnelly C.F. and Calver M.C. (2005) The abundance of a threatened arboreal marsupial in relation to anthropogenic disturbances at local and landscape scales in Mediterranean-type forests in south-western Australia. *Biological Conservation* **127**, 463-476.

Wegener A. (1984) Survey of animal species killed by the Ash Wednesday fires along the Great Ocean Road. *Geelong Naturalist* **21**, 13-16.

Williams J.E., Whelan R.J. and Gill A.M. (1994) Fire and environmental heterogeneity in southern temperate forest ecosystems: implications for management. *Australian Journal of Botany* **42**, 125-137.

Wilson B.A. (1996) Fire effects on vertebrate fauna and implications for fire management and conservation. In '*Fire and Biodiversity. The Effects and Effectiveness of Fire Management*'. Proceedings of the Conference held 8-9 October 1994, Footscray, Melbourne. (Department of the Environment Sport and Territories, Canberra)

Wilson B.A. and Friend G.R. (1999) Responses of Australian mammals to disturbance: a review. *Australian Mammalogy* **21**, 87-105.

Woinarski J.C.Z. (1999) Fire and Australian birds: a review. In *Australia's Biodiversity - Responses to Fire: Plants, Birds and Invertebrates*. (Eds A.M. Gill, J.C.Z. Woinarski and A. York). (Department of the Environment and Heritage, Canberra)

Woinarski J.C.Z. and Cullen J.M. (1984) Distribution of invertebrates on foliage in forests of south-eastern Australia. *Australian Journal of Ecology* **9**, 207-232.

Woinarski J.C.Z. and Recher H.F. (1997) Impact and response: a review of the effects of fire on the Australian avifauna. *Pacific Conservation Biology* **3**, 183-205.

Woodgate P. and Black P. (1988) *Forest Cover Changes in Victoria 1869-1987*. Department of Conservation, Forests and Lands, Melbourne.

York A. (1996) Long-term effects of fuel reduction burning on invertebrates in a dry sclerophyll forest. In '*Fire and Biodiversity. The Effects and Effectiveness of Fire Management*'. Proceedings of the Conference held 8-9 October 1994, Footscray, Melbourne. (Department of the Environment Sport and Territories, Canberra)

York A. (1999) Long-term effects of repeated prescribed burning on forest invertebrates: management implications for the conservation of biodiversity. In *Australia's Biodiversity - Responses to Fire. Plants, Birds and Invertebrates. Environment Australia Biodiversity Technical Paper No 1.* (Eds A.M. Gill, J.C.Z. Woinarski and A. York) pp. 181-266)

York A. (2000) Long-term effects of frequent low-intensity burning on ant communities in coastal blackbutt forests of southeastern Australia. *Austral Ecology* **25**, 83-98.

APPENDIX 1 - RESULTS OF WORKSHOP

Held 20th June 2007, Bendigo Pottery, Midland Highway, Epsom.

Attendees:

Geoff Park, North Central Catchment Management Authority (Facilitator)
Aaron Gay, North Central Catchment Management Authority
Lyndall Rowley, North Central Catchment Management Authority
Paula Camenzuli, North Central Catchment Management Authority
Ian Higgins, North Central Catchment Management Authority
Adrian Martins, North Central Catchment Management Authority
Arn Tolsma, Arthur Rylah Institute
David Cheal, Arthur Rylah Institute
Greg Horrocks, Monash University
David Baker-Gabb, Bush Heritage
Jeroen Van Veen, Country Fire Authority & Bush Heritage
Les Vearing, Department of Sustainability and Environment
Deanna Marshall, Department of Sustainability and Environment
Peter Johnson, Department of Sustainability and Environment
Mark Gilmore, Country Fire Authority
Roy Speechley, Parks Victoria
David Major, Parks Victoria
Cliff Ireland, Parks Victoria
Garry Cheers, Maryborough Field Naturalists
Ern Perkins, Castlemaine Field Naturalists
Leslie Perkins, Castlemaine Field Naturalists
Sabine Wilkens, La Trobe University & Bendigo Field Naturalists

Background and Aim

There is a growing awareness of the role of fire for ecological purposes, and that particular fire regimes are necessary for the maintenance of biodiversity.

Some have suggested that a restoration of aboriginal burning practices would greatly assist biodiversity maintenance in Box-Ironbark. Others suggest that these forests were not deliberately burnt. Little direct evidence exists either way, and what evidence there is relates to landscape-wide observations that cannot be referred to specific sites. In any event, given the dramatic changes in the landscape since European settlement, the ecological consequences of applying a burning regime in Box-Ironbark forests as they exist today have been inadequately researched.

Given the lack of pertinent data, landowners and land managers are perhaps our best source of information relating to the short or long-term effects of fire in these landscapes, and land manager input was considered critical to the current project. This workshop was therefore convened with the aim of synthesising that undocumented knowledge.

Attendees were divided into two independent groups of similar size, and asked to discuss a number of issues. These were:

- Likely aboriginal burning practices.
- Direct and indirect impacts of ecological burning on vegetation and habitat for fauna.
- Landholder or land manager observations on the effects of fire, or lack of it.
- The roles of fire in the restoration or regeneration of native vegetation in Box-Ironbark forests.

A brief introduction to each of these issues was given, and a number of specific questions was provided to guide the free-form round-table discussions.

Note that the views expressed are those of the attendees, recorded during the discussions. These views have not been edited for content, and should not be taken as representing the views of the authors, nor the CMA, nor the organisations whose members were at the workshop. The authors make no assertions in regard to content.

Pre-settlement Burning Practices

- The discussion of pre-European fire regimes may have been hampered by the absence of the indigenous representatives that had been invited to the workshop. Information was anecdotal, based on readings of historical records or previous conversations with relevant people, and merely reinforced the difficulty in interpreting the traditional role or extent of fire in the landscape.
- There is no doubt that the region in and around the Box-Ironbark forests was extensively used. Aborigines camped in several places regularly, such as grassy woodlands, moving closer to water sources in summer. Open, grassy areas were dug over extensively to harvest the roots of species such as Yam Daisy.
- The pattern of occupation would drive the fire regime. Burning of particular areas was probably done to promote food crops, attract animals, or reduce potential future fire intensity. Fire was also used for camp fires and smoke signals.
- However, much of the traditional knowledge relating to the use of fire in these ecosystems appears to have gone. Local leaders cannot tell us about the original use of fire, but have expressed doubt that it occurred in Box-Ironbark

forests.

- Writings and diary entries from early explorers mentioned fire in a generic sense, but did not give the impression that they saw large fires.
- George Augustus Robinson, the first Chief Protector for Aborigines in the Port Phillip District, provided no evidence of aboriginal burning despite numerous trips to the western districts.
- Environment Conservation Council references to aboriginal burning could not be supported by solid evidence and were not supported when challenged.
- Very difficult to get site-specific information, that is, the actual vegetation type in which fires were noted.
- There are references to fire in grassy or grassy-woodland areas in the explorers' reports, but not to forests. However, grassy woodlands are very different to Box-Ironbark, and if the latter were burnt it would undoubtedly have been at a lower frequency, and probably in autumn. Even if particular vegetation types were deliberately burnt, we have no information on the timing or frequency of burns, or the methods used to control those burns.
- Some have suggested that the forests were perhaps very different at settlement, but the Environment Assessment Council statements that the Box-Ironbark forests were more grassy and open were not adequately supported. Some early explorers spoke about thick scrub as well as open woodland.
- In any event, grassiness can be, and is maintained without fire, such as at Terrick Terrick.
- Ignition sources are naturally present in Box-Ironbark forests. The sources of ignition included lightning (around seven natural fires per year around Bendigo, and around 25-35 lightning fires in the Box-Ironbark region per year at present). This implies an average rate of around 100 lightning fires each year in Box-Ironbark forests in pre-European times (allowing for post-European clearing). It is likely that occasional escapes from small camp fires also occurred.
- Differential fire régimes probably existed across the region, with several ranges in the study area (for example, the Pyrenees) far more prone to lightning strikes than others. Forests around St Arnaud are often hit first during storms coming from the west. Lightning strikes are historically less frequent north of Castlemaine (the ranges deflect the storm clouds?).
- Large Box-Ironbark fires probably infrequent and occasional. In the Grampians, only one in four lightning strikes leads to a large fire.
- The rate of fire spread in Box-Ironbark is largely unknown. Fire is thought to travel 8-10 times faster in grassland. Extent of fire would depend on many factors, such as fire conditions and rainfall. Pre-settlement structure would have been different, with well-developed gully vegetation and wetlands, and this wet/dry patchiness would have affected the pattern of fire spread.
- Ironbark trees are considered less flammable, because of the bark type, but stringybark is more flammable (for example, in the forests south of

Castlemaine). When the canopy consists of Box and/or Ironbark species, there tends to be little spotting. However, Stringybark is more flammable and will spot. A common view that Box-Ironbark won't readily carry fire is considered incorrect.

- Mallee vegetation types within the Box-Ironbark ecosystems probably burnt more frequently. River Red-gum was possibly burnt deliberately, to prevent the build-up of heavy fuel from leading to hot fires.
- It is reasonable to infer burning régimes from plant responses.
- We have a forest type that recovers well from disturbance (sometimes assumed to be made up of the plants "we have left"). However, this is not necessarily a result of deliberate, applied fire.
- Note the Tertiary origin of fire-sensitive species, plant species that would have been unlikely to persist under a regular fire regime.
- It is doubtful that there was an intricate or sophisticated (applied) fire régime in Box-Ironbark forest before European settlement.

21st Century Landscape Changes

- There was general agreement that the Box-Ironbark forests had changed substantially since European settlement.
- North Harcourt gullies contained tree ferns in the 1850s, now lost (reference to sermon from "Happy Valley" near Castlemaine, under a tree-fern).
- We have relatively young forests now. Almost all the extant forest has been cut down for mining or other purposes around 2 to 4 times, and at the turn of the twentieth century there was little forest left in mined areas.
- Since the 1940s there has been an increase in the area under trees and a higher density of trees on public land. Most trees are small compared to those that were once there.
- During the 1950s to 1980s there was extensive thinning in Box-Ironbark forests (data showed the density was over 20 000 trees per hectare by then).
- Box-Ironbark forest is now seen by some in the community as a wildfire hazard, particularly if fuel loads increase after a wet year.
- Suggestion was made that vigorous shrub growth might inhibit large tree growth.
- Climate has warmed and dried over last 30-50 years, affecting fuel characteristics.
- Only a small proportion of fires now are a result of lightning.
- The timing of fires and fire intensity might be different now because of changes in species and structure. Summer burns are hot, autumn burns are cooler.
- For much of the time since European settlement there has been a policy of fire suppression. Fires from lightning strikes are generally extinguished rapidly.

- Fire may be absent now for long periods. For example, a landholder at Wychitella claims that there has been no fire recorded there for around 85 years.
- Coarse woody debris is seen as a wasted firewood resource by the public and thus there has been a generalized loss through collection of it in many areas. However, more coarse woody debris has been allowed to accumulate on some public land over the last 20 years because of tighter harvesting controls. However, large logs are often missing from the coarse woody debris in all forests.
- On private land, coarse woody debris still tends to be removed very early (seen as a fire hazard or an eye-sore). Small blocks are characterized by low to no coarse woody debris and high grazing intensity.
- Fuel build-up is perhaps slower now in some areas, and where fuel-reduction burning was once undertaken on an 8-10 year rotation it is now carried out on a 14-16 year rotation. However, leaf litter can build up quicker during severe drought years. Litter will also build up rapidly after fire, but levels plateau after a few years.
- Previous timber harvesting was associated with the creation of 'hot spots', where the heads were first piled together prior to burning. This would create a mosaic of soil and vegetation conditions different to that expected from a natural fire.
- There seems to be higher macropod abundance now, such that browsing and grazing have become issues in terms of plant survival and recruitment.
- Grassy ecosystems are highly modified and lack ability to fully regenerate.

Effects of Fires in Box-Ironbark

- There was no single answer to the question of weed response to fire, as the response was dependent on species, landscape position and management before and after the fire. In general, Box-Ironbark forests don't suffer from high levels of weed infestation, except in wetter, more fertile areas or where disturbance occurs. Common weeds are often annual grasses such as Hair-grass (*Aira* spp.) or Quaking Grass (*Briza* spp.), which have little influence on fire fuel hazard.
- There is usually little effect of fire on weeds on the drier ridges and upper slopes of Box-Ironbark or Heathy Dry forests. However, on lower slopes and gullies with better soils (such as those dominated by Yellow Box, Grey Box and Yellow Gum) there may be a marked increase in weeds. Increases were noted in the abundance of Spear Thistle (*Cirsium vulgare*), Cape Weed (*Arctotheca calendula*) and Cootamundra Wattle (*Acacia baileyana*) after the 1985 fire around Paddys Ranges. Such increases in weeds in gullies, particularly of shrubby species, will increase fuel loads.
- Pre-fire history and proximity to farmland are also major influences on the response by weeds. A hotter fire might open up the canopy more, and make an area more prone to windblown seed.

- Some weeds will eventually thin out as the forest ages (for example, Spear Thistle).
- Montpellier Broom (*Genista monspessulana*) is a major weed and does very well after fires, as does Gorse (*Ulex europaeus*). However, 'follow-up' burning might help eradicate these. Funding for weed control should (and does) concentrate on these landscape changers.
- Patches with a higher perimeter to area ratio are more exposed to the weed problem, particularly those at the interface between native bush and farmland. Horticultural weeds are a problem close to built-up areas because of garden escapes.
- Roads are a focus for weed problems (even tracks through native bush) because of run-off and nutrient accumulation. This can lead to higher fuel loads on roadsides (and thus also give a biased impression of forest fuel levels).
- Burning can lead to a blow out in the population of annual weeds. Hair-grass (*Aira* spp.) and Quaking Grass (*Briza* spp.) proliferate after fire, although they are also abundant after other disturbance to the soil. If burning is stopped, the native species eventually come to dominate again.
- Paradoxically, burning at an appropriate time can be a weed control procedure. For example, for the control of Canary Grass (*Phalaris*).
- The intensity of burn has an impact on weed response, so perhaps high intensity fire has different impacts that could be used for weed control.
- Some Orchids (e.g. Wax-lip) show flushes after fire, such as seen near Castlemaine. However, summer-flowering species might be badly affected depending on the season of burn. The endangered Yellow-lip Spider-orchid (*Caladenia xanthochila*) has not been seen since fire burnt some of its habitat near St Arnaud. In contrast, the absence of some Orchid species near Castlemaine may be due to a lack of fire.
- Lily and daisy species increase in abundance after fire, probably because of the nutrient pulse and opening up of the canopy. Milkmaids (*Burchardia umbellata*) increased substantially after fire in wetter forest near Castlemaine.
- *Poa* tussocks will regenerate rapidly after fire, and *Poa* was noted to increase near Castlemaine.
- Medium to high intensity fires encourage the native legumes, with their hard, long-lived seeds. For example, Golden Wattle (*Acacia pycnantha*), Spreading Wattle (*Acacia genistifolia*) and Hedge Wattle (*Acacia paradoxa*) respond very positively to fires. This was noted after the 1985 wildfire in Paddys Ranges, where hot fires removed the canopy over most of the park. Bush-peas (*Pultenaea* spp.) also increased markedly. However, cooler burns have far less effect on the Wattles and other shrubs, as noted in the Pyrenees Ranges.
- *Hakea* will regenerate well after fire, although they will also recruit to an extent after hot days. Daphne Heath (*Brachyloma daphnoides*) and Beard-heath (*Leucopogon* spp.) are thought to drop out after a long period without fire. In contrast, species such as Honey Pots (*Acrotriche serrulata*) and

Cranberry Heath (*Astroloma humifusum*) are able to persist without fire.

- Shrub promotion by fire has not always been seen after fires in Box-Ironbark forest, because local factors such as the soil seed bank will have a major effect, as will the degree of canopy removal. Vegetation communities are not totally stable, even without fire. Nonetheless, fire can be expected to change the structure and composition of native plant communities at least in the short-medium term.
- Conditions after fire have a large impact on the fate of shrub recruitment. Mortality is high if the forests remain dry, but good weather will result in a growth flush. Seasonal factors can also influence shrub recruitment. Photographs were provided of a bush block at Wattle Flat that was burnt in spring 1998, with little understorey after five years due to drought and browsing. In contrast, a small unburnt patch still contained dense *Acacia* and *Cassinia*.
- Sequential fires will alter post-fire response if the seed bank is depleted, and plants have been unable to grow to reproductive maturity.
- Burning at times of high environmental stress might give different (worse) results in terms of biodiversity outcomes. Burning prior to seed maturation will impact on soil seed bank.
- Sapling eucalypts mimic the shrub layer by increasing in abundance after fire, and may benefit some birds, suggesting that recruitment after fire can have positive benefits. In wetter areas near Castlemaine there was a substantial increase in the recruitment of Broad-leaved Peppermint (*Eucalyptus dives*) after fire. However, autumn burns might destroy eucalypt recruits or coppice growth.
- Some species decline after fire, for example, Cat's Claw Grevillea (*Grevillea alpina*). Several acres of dense *Cryptandra* in the Maryborough area were wiped out by a single cool burn some years ago, and the endangered Whipstick Westringia (*Westringia crassifolia*) is considered fire-sensitive.
- The current aim is not to burn big old trees because they can be susceptible to collapse after fire. This tendency can depend on the species, with scars on Yellow Gum (*Eucalyptus leucoxylon*) and readily-burnt bark of Yellow Box (*Eucalyptus melliodora*) increasing their susceptibility to collapse after fire. Coppice regrowth on badly burnt stumps may also eventually result in stump collapse. Trees over a certain size are protected from burning on public land, as they are in short supply and provide vital hollows for fauna.
- Macropod browsing appears to be having a major impact on Box-ironbark forests now, although the effects are likely to be exacerbated by long-term drought. The bigger macropods (Eastern Grey Kangaroo, Western Grey Kangaroo and Black Wallaby) are now at greatly increased abundance than previously – this browsing is intense enough to remove palatable shrub species from the landscape (including seed sources). Orchids in particular are susceptible to the removal of seed sources.
- However, lack of grazing in small blocks can lead to shrub thickening.

- Burning has negligible effects on macropod densities, which are high in any event – wallabies eat shrubs and occur in dense vegetation patches, while kangaroos occur in more open country and eat grasses. Nonetheless, grazing and browsing, including that by rabbits, can be high post-fire, and fenced exclusion plots have displayed higher species abundance and higher recruitment levels.
- Macropod densities and browsing are very separate issues to fire management.
- Beekeepers ask land managers not to burn but there is very little evidence to support their concerns. Presumably there can be physical injury to trees during fires, especially to those in or about to come into flower, with effects lasting for 2-3 years. Fires can delay flowering that beekeepers depend on. Swift Parrots are also dependent on flowering.
- Indicator organisms (animals) could be Swift Parrots, Grey-crowned Babblers, Tuans, Barking Owls, Regent Honeyeaters, Sugar Gliders, Brown Tree-creepers, Hooded Robins, Scarlet Robins, White-browed Babblers, Jacky Winters or Noisy Miners. These all require particular components of the forest.
- Effects of fire on fauna can be direct or indirect, and is strongly related to the season of burn. For example, spring fires will burn nests of birds such as Wrens and interfere with breeding. Conversely, autumn fires will impact on habitat. Similarly, spring fires will interfere with the reproduction of Brush-tailed Phascogales, while autumn fires will interfere with their seasonal movements (that require shelter).
- Concern was expressed about the reduction of habitat and food sources for ground dwelling birds, mammals, reptiles and invertebrates. However, the effects of fire on the flora structure may not necessarily extrapolate to the effects on fauna, and *vice-versa*.
- Over time, birds may be promoted by fires that open up the area, particularly if there is promotion of coarse woody debris. An example was given of a small, 2 ha site in which bird counts originally averaged 1.5 birds per visit, mostly Magpies or Galahs. Since a spring 1998 burn, birds have increased, and species such as Quails, Speckled Warblers and Rose Robins have been seen.
- An example was given of an area with big old trees, dense wattles and many logs. Kangaroo densities are sometime 70-80 per square km, and goannas are present. This represents an area where fire would have a severe impact on habitat.
- Possums and Koalas appear to decline after fire, then increase again in long-unburnt forest. Sugar Gliders and Possums will utilise Wattles in autumn and winter, and removal of the middle storey by fire would impact on them indirectly. Powerful Owls, Possums and Antechinus benefit from having long-undisturbed forest.
- Some attendees believed that ground foraging birds such as some Whistlers or Diamond Firetail may be affected in the short term by disturbance, but benefit in the long term. Conversely, others thought that fire might disadvantage birds such as Treecreepers.

- Reptile diversity tends to be low in Box-Ironbark, and fire that affects the ground layer may impact on them disproportionately. The endangered Pink-tailed Worm-Lizard lives near ant colonies. Will this be negatively affected by fire?
- Trees with hollows need to be protected from fire, as loss of hollows is considered to be one of the greatest threats to fauna.
- Fire will always impact on fauna in some way. We need to look at the long-term picture, and what eventually happens at a community or landscape level in terms of promoting fauna or otherwise. Spatial and temporal scales are important, as is the ability of particular species to recolonise.
- The isolation of private remnants may represent a barrier to recolonisation, and the impact of fire on fauna in these remnants may be more severe and long-lasting than in larger blocks.

Long-term Effects of Changed Fire Régimes

- Most weeds are relatively immune to fire age as a general statement, with infestations often linked to soil disturbance.
- Some weed species such as Montpellier Broom (*Genista monspessulana*) will increase rapidly after fire, but then continue to increase slowly even after a relatively long fire-free period. In contrast, Spear Thistle (*Cirsium vulgare*) decreases in the years after the fire.
- Fuel reaches a plateau very quickly, and decomposition appears to be faster than appreciated currently. Fine fuel build-up from absence of fire can be confounded with litter build-up from drought.
- Increase in fuels tends to be faster in some areas (for example, under Yellow Gum (*Eucalyptus leucoxylon*), or under Red Stringybark (*E. macrorhyncha*) with its greater grassiness). "Pencil-sized" fuel represents the greatest fire hazard.
- After a hot fire there may be a great increase in legumes (for example, Golden Wattle or Hedge Wattle) to relatively high levels but these shrubs retreat back to a ‘natural’ plateau over 20 years or so. For example, these species increased dramatically in parts of Paddys Ranges after a wildfire in 1985, but Hedge Wattle has mostly died back already, and Golden Wattle is in an advanced state of senescence. However, it can be difficult to differentiate the extent of natural post-fire senescence with that caused by drought. Golden Wattle is not seen as a fire hazard.
- As time since fire increases, we can expect a simplification of forest structure. However, this varies substantially between sites depending on grazing history and other factors. Two sites with similar overstorey and landscape position can have very different understorey compositions. Around Wedderburn and Fryers Ranges there are dense shrubs in the absence of fire. Soil type can influence response.
- Summer rain pulses produce a grassy fuel that raises the fire risk, and

subsequent fires may lead to the permanent loss of plant species. This can then lead indirectly to the permanent loss of fauna species. For example, a stand of tall Totem Poles (*Melaleuca decussata*) failed to recover for some reason after a fire in the Whipstick near Bendigo. Consequently, Grey-crowned Babblers disappeared.

- The control of grazing has led to increased fire risk in some areas (for example, Kooyoora State Park).
- We know little about the interaction between fuels and grazing in Box-Ironbark forest.
- Erosion of shallow, infertile soils on slopes and the washing down-slope of litter and ash may be a problem after fire, and the long-term consequences could be a reduction in soil fertility. Some Red Ironbark ridges are still bare after the 1985 Paddys Ranges wildfire.

Is There a Need to Burn Box-Ironbark?

- Any justification for ecological burning must be clearly separated from that for fuel-reduction burning. However, there was a suggestion that burning of buffer strips might sometimes be appropriate in order to protect more fire-sensitive vegetation, blurring the distinction between the two burn types.
- In some situations fire may be needed for regeneration of some species. For example, tiny fires are used to promote microhabitat for Orchids. Totem Poles (*Melaleuca decussata*) might need occasional fire to persist.
- Keep in mind that many species occur at any one site and in any one vegetation community, and there is no single fire regime that suits all species.
- A background level of fire will occur naturally, without applying additional burning.
- As a general statement, fires are not necessary in Box-Ironbark (even though the use of fire is sometimes assumed). Low levels of recruitment of most canopy and shrub species occur in the absence of fire, depending on factors such as spring rain. In Box-Ironbark forest around Castlemaine and Maryborough, Wattle recruits are always present. However, Red Ironbark is considered to be a poor recruiter from seeds, even after fire, unless conditions are suitable. Seasonal variation occurs. For example, seed production by Wattles in research at Reef Hills last year was low (less than 400 per hectare).
- As a general statement, we <u>can</u> burn without undue ecological damage, but there are local exceptions depending on specific values. For example, we may be able to use fire at a particular time of the year to reduce annual weeds, or exclude fire to protect large old trees or fire-sensitive species.
- The ideal fire régime is determined by the desired landscape pattern, and that is determined by other over-arching concerns, notably including grazing and weeds (in the smaller blocks), other disturbances, plant life histories, and plant recruitment and senescence stages.
- The seasonality and frequency of fire are important considerations in

determining the effects of fire on flora, fauna or fauna habitat.

- It was considered doubtful that any animals would disappear through a lack of fire, and few plant species (except for perhaps some isolated rare ones). Box-Ironbark doesn't appear to have the succession-dependent fauna species that are adapted to a changing environment, such as Smoky Mouse (from heathlands).
- Climate Change is an over-arching consideration that is possibly changing the response of plant species (note the example of Whirrakee Wattle (*Acacia williamsonii*) failing to regenerate because of drought). If the forests continue to dry out, then historical or current fire regimes become largely irrelevant.
- There is generally a lower amount of litter accumulation with age in Box-Ironbark forest compared to other forest types. An estimated 7-15 years is required to build up fuel sufficiently to allow a good burn. However, despite these forests being almost semi-arid, they will still burn very hot if conditions allow, even with little fuel on the ground.
- Overall fuel loads may be very different now to what they were before European settlement. We have no way of knowing whether any fire now would be 'natural' in terms of fire intensity or response.
- There is more age-based accumulation of shrubs in the southern Box-Ironbark forests (for example, some Heathy Dry Forest around Castlemaine). Conversely, some areas with high species richness had a relatively low fuel load.
- Fires burn on different fuels (for example, in *E. macrorhyncha* on grass tussocks; in *E. leucoxylon* on the discarded bark from large trees, in Red Ironbark *E. tricarpa* on tree leaf litter), and all these fuels have different levels of flammability.
- Fuel differentials are too unreliable to use for controlling unbounded burns. That is, there is no guarantee that the fire will stop at a different vegetation type.
- There is evidence of the adverse impacts of some fires. Some species in Box-Ironbark forests are fire-sensitive, and may be eliminated by a single burn.
- Maryborough Field Naturalists' Club consider that burning would result in a decline in the cover of fire-sensitive species, changes in the composition and structure of native communities, reduction of habitat and food sources for ground-dwelling birds, mammals, reptiles and invertebrates, and weed invasion.
- Paradoxically, occasional fire may be needed to promote hollows in trees. It is believed that the development of hollows in old Box or Ironbark trees does not necessarily come about from the process of ageing in itself, but results from damage to the bark and outer layers by fire or wind storm. Examples were given of unburnt logging coupes where trees 50-70 cm DBH had almost no hollows, and burnt areas where most trees over 25 cm DBH contained them.
- Small blocks have value for connectivity despite lower levels of coarse woody debris, and tend to be excluded from prescribed burns.

- Small grassy blocks do get some consideration (but consideration only) for prescribed burns, as they may contain species more closely affiliated with grasslands (for example, Kangaroo Grass (*Themeda triandra*)). Grassy Woodlands are hard to burn from a practical perspective.
- The sacrifice of certain strategic patches of vegetation by fuel-reduction burning was considered to be a tolerable approach to preserving broader landscapes. This already happens around towns such as Castlemaine.
- We still have little knowledge of historical fire regimes in Box-Ironbark forest, appropriate present-day regimes or the differential effects of burning.

Further Research

- Bring the fire mapping together (back to the 1980s). Pull together any available data from fuel-reduction burning.
- Compile a list of potential research projects, including experimental management.
- Money & resources are needed to study the impacts of fires. This can best be done using pre- and post-fire data from fuel reduction burns undertaken in future in appropriate forest types.
- Model the possible spread of fire in Box-Ironbark.
- Core sampling of relatively undisturbed gullies or alluvial terraces might locate layers of ash washed off slopes, and help document historical fire frequency.
- Dendrochronology of grass trees might be useful for determining fire regimes, although doubt that they are old enough.
- What are we actually trying to preserve? What is the forest type and structure that we want? Is fire the most appropriate way to achieve this, or are there other means such as thinning and macropod control?
- More research on animals and habitat interactions is needed, for example, Noisy Miners.
- Field sites for such work exist and are known.

Post-workshop Feedback

Interested persons who had been unable to attend the workshop were invited to make comments via email. Two emails were received, from Tim Kelly (River Restorations), and Greg Turner (Department of Primary Industry). Mark Gilmore (Country Fire Authority) and Lyndall Rowley (North Central CMA) provided further comments following their attendance at the workshop.

- Vegetation has changed over the last 5 million years, from rainforest to fire-adapted species. Drying and burning is now a self-perpetuating cycle.

- During storms, it is often too wet in Box-Ironbark for lightning to start a fire. This will likely change as the climate gets hotter and drier. Many fires caused by lightning strikes are in drier grasslands.
- Box-Ironbark experiences a wide range of fire intensities, but regeneration occurs well from all fire regimes. This variation also leads to great variation in plant species.
- With constant vegetation regeneration and 'cloning', flora change in Box-Ironbark is more likely to be driven by events such as drought than by fire.
- Stringybark burns more often and has a grassy understorey, but its location on sloping ground probably has as much to do with fire intensity as weather or vegetation type.
- Aborigines burnt grassland areas in winter to achieve a cool burn.
- Forests have been burnt and cleared substantially since the 1800s, and grazing pressure has been high at times. From 1960 to 1990 forests were thinned and burnt at around 10 year rotations.
- Native grasslands require burning to prevent grassy swards "choking".
- Rough Spear-grass (*Austrostipa scabra*) is not robust enough to persist after a hot fire.
- Late autumn burn of revegetation area dominated by Wirilda (*Acacia retinodes*), with no follow-up rain, resulted in high plant mortality.
- Should not burn at all in dry seasons, as it could lead to an understorey "wasteland".
- Malleefowl (*Leipoa ocellata*) are known to prefer long-unburnt habitat.
- Effective fire buffers would destroy too much habitat, and in any event be ineffective on days of high fire-danger.
- Ability to manipulate vegetation in reserves less than 1000 ha to prevent wildfire is small, as buffer area becomes proportionately larger for small reserves.
- Effects of fire on birds will depend on the connectivity of a site. For example, there is a strong probability that the local extinction of an isolated colony of Shy Heathwren (*Anguilla australis*) was due to a poorly-planned fuel reduction burn that removed shrub thickets.
- Powerful Owls (*Ninox strenua*) are assumed to prefer long-undisturbed forest, yet high numbers have been observed in Lerderderg forest recovering from fire.
- Outcomes and objectives for burning must be clearly articulated.
- Vegetation succession is an on-going process. Some aspects of the ecosystem have been changed forever, and we should not be trying to force the ecosystems back to some 'ideal' state.
- Any fire-sensitive species would be "long gone", so management decisions should not worry about past conditions.

- Need to capture land management histories before the information disappears. This would give us a better picture of the possible responses to further management action.
- DSE fire management has good records of fuel reduction burns, and these would facilitate field work to determine the effects of fire, and the optimal fire regime.
- We need a step-by-step, evidence-based approach to fire management.
- Fire effects may be over-ridden by other processes.

APPENDIX 2 - PLANT VITAL ATTRIBUTES FOR SOME BOX-IRONBARK SPECIES

Species have been sorted in order of their fidelity as Box-Ironbark species. Refer to Page 30 for key to symbols used.

FFG	EPBC	VROTS	Scientific Name	Common Name	BIB Ind	KFR	SEED	VEG	TIR	JUV	SPP LIFE	INDIV LIFE
			Acacia difformis	Drooping Wattle	C-	3	S		I	5	50	MP
		r	*Acacia flexifolia*	Bent-leaf Wattle	C-	-						
		r	*Austrostipa breviglumis*	Cane Spear-grass	C-	-						
			Bursaria spinosa subsp. lasiophylla	Hairy Bursaria	C-	3		V	T			
			Cheiranthera cyanea var. cyanea	Blue Finger-flower	C-	NS				5	50	MP
			Cyanicula caerulea	Blue Fairy	C-	NS		U	T	5	100	MP
			Dampiera dysantha	Shrubby Dampiera	C-	-						
			Daviesia benthamii subsp. humilis	Spiny Bitter-pea	C-	-						
			Dichelachne hirtella	Hairy Plume-grass	C-	-						
			Dillwynia ramosissima	Bushy Parrot-pea	C-	-						
			Eucalyptus tricarpa subsp. tricarpa	Red Ironbark	C-	-						
			Goodenia hederacea subsp. hederacea	Ivy Goodenia	C-	NS			T	2	100	MP
			Hyalosperma praecox	Mayweed Sunray	C-	-						
			Juncus remotiflorus	Diffuse Rush	C-	NS		V	T	2	100	
			Melichrus urceolatus	Urn Heath	C-	NS			I	5	100	LP
			Persoonia rigida	Hairy Geebung	C-	2	G		T	5	100	
			Pterostylis cycnocephala	Swan Greenhood	C-	NS		U	T	5	100	LP
			Pultenaea pedunculata	Matted Bush-pea	C-	-						
			Rhodanthe laevis	Smooth Sunray	C-	-						
			Stuartina muelleri	Spoon Cudweed	C-	NS	C	X	T	1	100	A
		v	*Thelymitra X macmillanii*	Crimson Sun-orchid	C-	NS		U	T	5	100	LP
		v	*Acacia ausfeldii*	Ausfeld's Wattle	R	-						
		r	*Boronia anemonifolia subsp. aurifodina*	Goldfield Boronia	R	-						
			Hibbertia exutiaces	Spiky Guinea-flower	R	-						
			Olearia teretifolia	Cypress Daisy-bush	R	-						
			Philotheca verrucosa	Fairy Wax-flower	R	-						

			Prostanthera denticulata	Rough Mint-bush	R	3			I	5	50	
			Pultenaea largiflorens	Twiggy Bush-pea	R	3			I	5	50	
			Xanthorrhoea glauca subsp. angustifolia	Grey Grass-tree	R	1		U	T		100	LP
			Xerochrysum viscosum	Shiny Everlasting	R	NS	G		T	2	100	SP
			Acacia aspera	Rough Wattle	R-	-						
		r	*Acacia williamsonii*	Whirrakee Wattle	R-	-						
		r	*Cassinia diminuta*	Dwarf Cassinia	R-	-						
f		r	*Eucalyptus froggattii*	Kamarooka Mallee	R-	-						
		r	*Eucalyptus polybractea*	Blue Mallee	R-	2	G	Y	I		100	LP
			Eucalyptus viridis	Green Mallee	R-	3	G	Y	I	10	100	LP
		r	*Grevillea dryophylla*	Goldfields Grevillea	R-	-						
		r	*Grevillea obtecta*	Fryerstown Grevillea	R-	-						
f	V	e	*Lepidium pseudopapillosum*	Erect Peppercress	R-	-						
		r	*Olearia tubuliflora*	Rayless Daisy-bush	R-	-						
f		v	*Phebalium festivum*	Dainty Phebalium	R-	2	G	X	I	5	50	MP
			Philotheca angustifolia	Narrow-leaf Wax-flower	R-	NS						
f	E	e	*Pterostylis despectans*	Lowly Greenhood	R-	-						
		r	*Pterostylis setifera*	Bristly Greenhood	R-	NS						
f		v	*Pultenaea graveolens*	Scented Bush-pea	U	3			I			MP
		r	*Swainsona behriana*	Southern Swainson-pea	U	2	Z	W	I	1		MP
f	E	e	*Westringia crassifolia*	Whipstick Westringia	U	-						
			Acacia acinacea s.l.	Gold-dust Wattle	W	-						
			Acacia acinacea s.s.	Gold-dust Wattle	W	-						
			Acacia genistifolia	Spreading Wattle	W	2	S		I	5	50	MP
			Acacia pycnantha	Golden Wattle	W	NS	S	X	I	5	100	MP
			Acrotriche serrulata	Honey-pots	W	2		Y	T	5	50	LP
			Amyema miquelii	Box Mistletoe	W	2	D		R	5	100	MP
			Arthropodium fimbriatum	Nodding Chocolate-lily	W	NS			T	1	100	MP
			Arthropodium strictum s.l.	Chocolate Lily	W	NS			T	2	100	MP
			Astroloma humifusum	Cranberry Heath	W	3	G	W	T	5	50	MP
			Austrodanthonia caespitosa	Common Wallaby-grass	W	-						
			Austrodanthonia eriantha	Hill Wallaby-grass	W	NS		Y	T	1	100	MP
			Austrodanthonia fulva	Copper-awned Wallaby-grass	W	NS					100	

			Austrodanthonia geniculata	Kneed Wallaby-grass	W	NS		Y	T	1	100	MP
			Austrodanthonia pilosa	Velvet Wallaby-grass	W	NS	S	Y	T	1	100	MP
			Austrodanthonia setacea	Bristly Wallaby-grass	W	NS		Y	T	1	100	MP
			Austrostipa mollis	Supple Spear-grass	W	NS		U	T	1	100	SP
			Brachyloma daphnoides	Daphne Heath	W	1	G	W	T	10	100	LP
			Brachyscome multifida	Cut-leaf Daisy	W			W				SP
			Brachyscome perpusilla	Rayless Daisy	W	NS	C	X	T	1	100	A
			Brunonia australis	Blue Pincushion	W	NS		U	T	2	50	SP
			Burchardia umbellata	Milkmaids	W	3		U				
			Caladenia cucullata	Hood Orchid	W	NS		U	T	5	100	LP
			Caladenia gracilis	Musk Hood-orchid	W	NS		U	T	5	100	LP
			Calochilus robertsonii	Purple Beard-orchid	W	NS		U	T	5	100	LP
			Calytrix tetragona	Common Fringe-myrtle	W	2	G	V	T	5	100	MP
			Cassinia arcuata	Drooping Cassinia	W	2	S		T	5	100	MP
			Chamaescilla corymbosa var. corymbosa	Blue Stars	W	NS		U		1	50	MP
			Cheilanthes austrotenuifolia	Green Rock-fern	W	NS	Z	V	T	2	100	LP
			Chrysocephalum apiculatum s.l.	Common Everlasting	W	3		V	T	2	50	ASP
			Chrysocephalum semipapposum	Clustered Everlasting	W	NS	S		T	2	50	
			Correa reflexa	Common Correa	W	3	G	Y	T	5		MP
			Corunastylis sp. aff. rufa (Goldfields)	Dark Midge-orchid	W	NS		U	T	5	100	LP
			Crassula decumbens var. decumbens	Spreading Crassula	W	NS	C		T	1	100	A
			Cymbonotus preissianus	Austral Bear's-ear	W	NS		U	T	2	50	SP
			Daviesia leptophylla	Narrow-leaf Bitter-pea	W	2			I	5	20	MP
			Daviesia ulicifolia	Gorse Bitter-pea	W	2		Y				
			Dianella revoluta s.l.	Black-anther Flax-lily	W	NS	Z	V	T	5	100	MP
			Dillwynia cinerascens s.l.	Grey Parrot-pea	W	2	S		I	5	50	MP
			Dillwynia sericea	Showy Parrot-pea	W	2			I	5	50	MP
			Diuris pardina	Leopard Orchid	W	NS		U	T	5	100	LP
			Einadia hastata	Saloop	W	NS	C/D		T	2	100	SP
			Einadia nutans subsp. nutans	Nodding Saltbush	W	NS	C		T	2	100	SP
			Elymus scaber var. scaber	Common Wheat-grass	W	NS		Y	T	1	100	MP
			Eucalyptus behriana	Bull Mallee	W	3	G	Y	I/T	10	100	LP
			Eutaxia microphylla var. diffusa	Spreading Eutaxia	W	3			K	5	50	
			Eutaxia microphylla var. microphylla	Common Eutaxia	W	-						

			Exocarpos cupressiformis	Cherry Ballart	W	3	G	Y	T	10	50	LP
			Gnaphalium indutum	Tiny Cudweed	W	-						
			Gompholobium huegelii	Common Wedge-pea	W	2		X	I	5	50	
			Gonocarpus elatus	Tall Raspwort	W	NS		V	R	2	100	SP
			Gonocarpus tetragynus	Common Raspwort	W	NS	G	W	T	2	100	SP
			Goodenia blackiana	Black's Goodenia	W	-						
			Goodenia geniculata	Bent Goodenia	W	2	G	V	I	2	100	MP
			Goodenia pinnatifida	Cut-leaf Goodenia	W	NS			T	2	100	MP
			Grevillea alpina	Cat's Claw Grevillea	W	NS	G		T	10	100	MP
			Hakea decurrens subsp. physocarpa	Bushy Needlewood	W	2	G	X	I	5	50	LP
			Hibbertia obtusifolia	Grey Guinea-flower	W	NS	G	V	T			MP
			Hibbertia riparia	Erect Guinea-flower	W	3	G	V	I	5	50	MP
			Hibbertia sericea s.l.	Silky Guinea-flower	W	3	G	Y	I	5	50	MP
			Hydrocotyle callicarpa	Small Pennywort	W	NS	S		T	1	100	A
			Hydrocotyle foveolata	Yellow Pennywort	W	-						
			Hydrocotyle laxiflora	Stinking Pennywort	W	NS	Z	Y	T	2	100	
			Hypoxis glabella var. glabella	Tiny Star	W	3	S	U	I	2	50	MP
			Isolepis marginata	Little Club-sedge	W	NS	G	X	T	1	100	A
			Joycea pallida	Silvertop Wallaby-grass	W	2	Z	V	T	2	100	LP
			Juncus amabilis	Hollow Rush	W	-						
			Juncus bufonius	Toad Rush	W	NS	S	X	T	1	100	A
			Juncus subsecundus	Finger Rush	W	NS		V	T	2	100	
			Lachnagrostis filiformis	Common Blown-grass	W	NS			T	1	100	ASP
			Lagenophora huegelii	Coarse Bottle-daisy	W	NS		U	T	1	100	
			Lepidosperma curtisiae	Little Sword-sedge	W	NS		Y	I	5	100	
			Leptomeria aphylla	Leafless Currant-bush	W	NS	S			10	100	LP
			Leptorhynchos tenuifolius	Wiry Buttons	W	-						
			Leptospermum myrsinoides	Heath Tea-tree	W	3		Y	I	5	100	MP
			Leucopogon fletcheri subsp. brevisepalus	Twin-flower Beard-heath	W	-						
			Leucopogon rufus	Ruddy Beard-heath	W	NS	C	W	T	5	100	MP
			Leucopogon virgatus	Common Beard-heath	W	-						MP
			Levenhookia dubia	Hairy Stylewort	W	NS	S		T	1	100	A
			Lomandra filiformis	Wattle Mat-rush	W	NS	Z	V	T	5	100	MP
			Lomandra multiflora subsp. multiflora	Many-flowered Mat-rush	W	NS		Y	I	5	100	MP

			Lomandra nana	Dwarf Mat-rush	W	NS		Y	I	5	100	MP
			Maireana enchylaenoides	Wingless Bluebush	W	-						
			Melaleuca uncinata	Broombush	W	NS	G	W	I	5	100	LP
			Microlaena stipoides var. stipoides	Weeping Grass	W	NS	Z	W	T	1	100	SP
			Microseris sp. 3	Yam Daisy	W	NS	G	W	T	2	100	SP
			Millotia tenuifolia var. tenuifolia	Soft Millotia	W	NS		X	T	1	100	A
			Myosotis australis	Austral Forget-me-not	W	NS	C		T	1	100	ASP
			Ozothamnus obcordatus	Grey Everlasting	W	-						
			Ozothamnus retusus	Rough Everlasting	W	-						
			Pelargonium rodneyanum	Magenta Stork's-bill	W	-						
			Pentapogon quadrifidus var. quadrifidus	Five-awned Spear-grass	W	NS	S	V	T	2	100	
			Pimelea humilis	Common Rice-flower	W	NS	G	Y		5	50	MP
			Pimelea linifolia	Slender Rice-flower	W	-						
			Poa sieberiana	Grey Tussock-grass	W	NS		V	T	2	100	LP
			Poranthera microphylla s.l.	Small Poranthera	W	NS	C		T	1	100	ASP
		r	*Pterostylis smaragdyna*	Emerald-lip Greenhood	W	NS			T			
			Pultenaea laxiflora	Loose-flower Bush-pea	W	3	S		I	5	50	
			Pultenaea prostrata	Silky Bush-pea	W	3	S		I	5	100	
			Ranunculus sessiliflorus	Annual Buttercup	W	NS	S		T	1	100	
			Rumex brownii	Slender Dock	W	NS	Z	V	T	1	100	SP
			Sebaea ovata	Yellow Sebaea	W	NS	S	X	T	1	100	A
			Solenogyne dominii	Smooth Solenogyne	W	NS		W	T	1	100	MP
			Stylidium graminifolium s.l.	Grass Triggerplant	W	NS		Y	T	2	100	SP
			Stylidium inundatum	Hundreds and Thousands	W	NS	S	X	T	1	100	A
			Tetratheca ciliata	Pink-bells	W	NS	G/S	Y	I	5	100	MP
			Themeda triandra	Kangaroo Grass	W	2	C	Y	T	2	100	LP
			Tricoryne elatior	Yellow Rush-lily	W	3		V	I	2	100	
			Veronica plebeia	Trailing Speedwell	W	NS		W	T	1	100	SP
			Wahlenbergia stricta subsp. stricta	Tall Bluebell	W	NS	G	W	T	2	100	SP

APPENDIX 3 - LIKELY EFFECT OF FIRE ON BIRD SPECIES

These data are the result of expert opinion from researchers at Arthur Rylah Institute, Monash University and Deakin University, and are best estimates of the likely response to low-intensity fire. Note that actual response to fire will vary substantially depending on factors such as timing, severity, extent, post-fire conditions and previous disturbance history. * introduced species, HN highly negative, SN slightly negative, HP highly positive, SP slightly positive, N neutral, U unknown.

Name		Nesting/Sheltering		Foraging		Impact of fire in short or long-term			
Common	Scientific	Ground	Low (< 2 m)	Ground	Shrub layer	Spring burn, short-term	Spring burn, long-term	Autumn burn, short-term	Autumn burn, long-term
Emu	*Dromaius novaehollandiae*	√		√		SN	HP	SP	HP
Brown Quail	*Coturnix ypsilophora*	√		√		SN	HP	SP	HP
Black-shouldered Kite	*Elanus axillaris*					SP	SP	SP	SP
Square-tailed Kite	*Lophoictinia isura*					SP	SP	SP	SP
Black Kite	*Milvus migrans*					SP	SP	SP	SP
Whistling Kite	*Haliastur shenurus*					SP	SP	SP	SP
Brown Goshawk	*Accipiter fasciatus*					SP	SP	SP	SP
Grey Goshawk	*Accipiter novaehollandiae*					SP	SP	SP	SP
Collared Sparrowhawk	*Accipiter cirrhocephalus*					SP	SP	SP	SP
Wedge-tailed Eagle	*Aquila audax*					SP	SP	SP	SP
Little Eagle	*Hieraaetus morphnoides*					SP	SP	SP	SP
Brown Falcon	*Falco berigora*					SP	SP	SP	SP
Australian Hobby	*Falco longipennis*					SP	SP	SP	SP
Grey Falcon	*Falco hypoleucos*					SP	SP	SP	SP
Black Falcon	*Falco subniger*					SP	SP	SP	SP
Peregrine Falcon	*Falco peregrinus*					SP	SP	SP	SP
Nankeen Kestrel	*Falco cenchroides*					SP	SP	SP	SP
Little Button-quail	*Turnix velox*	√		√		SN	HP	SP	HP
Painted Button-quail	*Turnix varia*	√		√		SN	HP	SP	HP

Bush Stone-curlew	*Burhinus grallarius*	√		√		HN	SN	SN	SN
Common Bronzewing	*Phaps chalcoptera*			√		SP	SP	SP	SP
Crested Pigeon	*Ocyphaps lophotes*			√		SP	SP	SP	SP
Diamond Dove	*Geopelia cuneata*			√		SP	SP	SP	SP
Peaceful Dove	*Geopelia striata*			√		SP	SP	SP	SP
Yellow-tailed Black-Cockatoo	*Calyptorhynchus funereus*					N	N	N	N
Gang-gang Cockatoo	*Callocephalon fimbriatum*					N	N	N	N
Galah	*Cacatua roseicapilla*			√		SP	SP	SP	SP
Long-billed Corella	*Cacatua tenuirostris*			√		SP	SP	SP	SP
Little Corella	*Cacatua sanguinea*			√		SP	SP	SP	SP
Sulphur-crested Cockatoo	*Cacatua galerita*			√		SP	SP	SP	SP
Cockatiel	*Nymphicus hollandicus*					SP	SP	SP	SP
Musk Lorikeet	*Glossopsitta concinna*					N	N	N	N
Little Lorikeet	*Glossopsitta pusilla*					N	N	N	N
Purple-crowned Lorikeet	*Glossopsitta porphyrocephala*					N	N	N	N
Superb Parrot	*Polytelis swainsonii*					N	N	N	N
Crimson Rosella	*Platycercus elegans*			√		SP	SP	SP	SP
Yellow Rosella	*Platycercus e. flaveolus*			√		SP	SP	SP	SP
Eastern Rosella	*Platycercus eximius*			√		SP	SP	SP	SP
Swift Parrot	*Lathamus discolor*					N	N	N	N
Red-rumped Parrot	*Psephotus haematonotus*			√		SP	SP	SP	SP
Blue-winged Parrot	*Neophema chrysostoma*			√		SP	SP	SP	SP
Turquoise Parrot	*Neophema pulchella*			√		SP	SP	SP	SP
Pallid Cuckoo	*Cuculus pallidus*					N	N	N	N
Brush Cuckoo	*Cacomantis variolosus*					N	N	N	N
Fan-tailed Cuckoo	*Cacomantis flabelliformis*					N	N	N	N
Black-eared Cuckoo	*Chrysococcyx osculans*					N	N	N	N
Horsfield's Bronze-Cuckoo	*Chrysococcyx basalis*					N	N	N	N
Shining Bronze-Cuckoo	*Chrysococcyx lucidus*					N	N	N	N
Powerful Owl	*Ninox strenua*					N	N	N	N
Barking Owl	*Ninox connivens*					N	N	N	N
Southern Boobook	*Ninox boobook*					N	N	N	N

Masked Owl	*Tyto novaehollandiae*					N	N	N	N
Barn Owl	*Tyto alba*					N	N	N	N
Tawny Frogmouth	*Podargus strigoides*					N	N	N	N
White-throated Nightjar	*Eurostopodus mystacalis*	√				SN	N	SN	N
Spotted Nightjar	*Eurostopodus argus*	√				SN	N	SN	N
Australian Owlet-nightjar	*Aegotheles cristatus*					N	N	N	N
White-throated Needletail	*Hirundapus caudacutus*					U	U	U	U
Fork-tailed Swift	*Apus pacificus*					U	U	U	U
Laughing Kookaburra	*Dacelo novaeguineae*					SP	SP	SP	SP
Sacred Kingfisher	*Todiramphus sanctus*					SP	SP	SP	SP
Rainbow Bee-eater	*Merops ornatus*	√				N	N	N	N
White-throated Treecreeper	*Cormobates leucophaeus*					N	N	N	N
Brown Treecreeper	*Climacteris picumnus*			√		HN	SN	HN	SN
Superb Fairy-wren	*Malurus cyaneus*		√	√	√	SN	SP	SN	SP
Spotted Pardalote	*Pardalotus punctatus*	√				SN	N	SN	N
Striated Pardalote	*Pardalotus striatus*					N	N	N	N
White-browed Scrubwren	*Sericornis frontalis*		√	√	√	SN	SP	SN	SP
Chestnut-rumped Heathwren	*Hylacola pyrrhopygia*		√	√	√	SN	SP	SN	SP
Speckled Warbler	*Chthonicola sagittata*		√	√	√	SN	N	SN	N
Weebill	*Cmicrornis brevirostris*		√		√	N	N	N	N
Western Gerygone	*Gerygone fusca*		√			N	N	N	N
Brown Thornbill	*Acanthiza pusilla*		√	√	√	SN	SP	SN	SP
Chestnut-rumped Thornbill	*Acanthiza uropygialis*		√			SN	SP	SN	SP
Buff-rumped Thornbill	*Acanthiza reguloides*		√	√	√	SN	SN	SN	SN
Yellow-rumped Thornbill	*Acanthiza chrysorrhoa*		√	√	√	SN	SN	SN	SN
Yellow Thornbill	*Acanthiza nana*					N	N	N	N
Striated Thornbill	*Acanthiza lineata*					N	N	N	N
Southern Whiteface	*Aphelocephala leucopsis*		√		√	N	N	N	N
Red Wattlebird	*Anthochaera carunculata*				√	N	SP	N	SP
Little Wattlebird	*Anthochaera chrysoptera*				√	N	SP	N	SP
Noisy Friarbird	*Philemon corniculatus*					N	N	N	N
Little Friarbird	*Philemon citreogularis*					N	N	N	N
Regent Honeyeater	*Xanthomyza phrygia*					N	N	N	N

Noisy Miner	*Manorina melanocephala*			√		SN	SP	SN	SP
Yellow-faced Honeyeater	*Lichenostomus chrysops*				√	N	SP	N	SP
White-eared Honeyeater	*Lichenostomus leucotis*				√	N	SP	N	SP
Yellow-tufted Honeyeater	*Lichenostomus melanops*				√	N	SP	N	SP
Fuscous Honeyeater	*Lichenostomus fuscus*				√	N	SP	N	SP
White-plumed Honeyeater	*Lichenostomus penicillatus*			√	√	N	SP	N	SP
Black-chinned Honeyeater	*Melithreptus gularis*					N	N	N	N
Brown-headed Honeyeater	*Melithreptus brevirostris*					N	N	N	N
White-naped Honeyeater	*Melithreptus lunatus*					N	N	N	N
Painted Honeyeater	*Grantiella pictus*					N	N	N	N
Crescent Honeyeater	*Phylidonyris pyrrhoptera*					N	SP	N	SP
New Holland Honeyeater	*Phylidonyris novaehollandiae*		√		√	SN	SN	SN	SN
Tawny-crowned Honeyeater	*Phylidonyris melanops*		√		√	SN	SP	SN	SP
Eastern Spinebill	*Acanthorhynchus tenuirostris*		√		√	SN	SP	SN	SP
White-fronted Chat	*Epthianura albifrons*		√	√	√	SN	SP	SN	SP
Jacky Winter	*Microeca fascinans*					SN	N	SN	N
Scarlet Robin	*Petroica multicolor*		√	√	√	SN	SN	SN	SN
Red-capped Robin	*Petroica goodenovii*		√	√	√	SN	SN	SN	SN
Flame Robin	*Petroica phoenicea*		√	√	√	SN	SN	SN	SN
Hooded Robin	*Melanodryas cucullata*				√	SN	SN	SN	SN
Eastern Yellow Robin	*Eopsaltria australis*		√	√	√	SN	SP	SN	SP
Grey-crowned Babbler	*Pomatostomus temporalis*		√	√	√	SN	SP	SN	SP
White-browed Babbler	*Pomatostomus superciliosus*		√	√	√	SN	SP	SN	SP
Spotted Quail-thrush	*Cinclosoma punctatum*	√		√	√	SN	N	SN	N
Varied Sittella	*Daphoenositta chrysoptera*					N	N	N	N
Crested Shrike-tit	*Falcunculus frontatus*					N	N	N	N
Crested Bellbird	*Oreoica gutturalis*		√		√	SN	SP	SN	SP
Gilbert's Whistler	*Pachycephala inornata*		√		√	SN	SP	SN	SP
Golden Whistler	*Pachycephala pectoralis*		√		√	SN	SP	SN	SP
Rufous Whistler	*Pachycephala rufiventris*		√		√	SN	SP	SN	SP
Grey Shrike-thrush	*Colluricincla harmonica*		√	√	√	SN	SP	SN	SP

Leaden Flycatcher	*Myiagra rubecula*					N	N	N	N
Satin Flycatcher	*Myiagra cyanoleuca*					N	N	N	N
Restless Flycatcher	*Myiagra inquieta*					N	N	N	N
Magpie-lark	*Grallina cyanoleuca*			√		SN	N	SN	N
Rufous Fantail	*Rhipidura rufifrons*		√			SN	N	N	N
Grey Fantail	*Rhipidura fuliginosa*		√			SN	N	N	N
Willie Wagtail	*Rhipidura leucophrys*		√	√		SN	N	SN	N
Black-faced Cuckoo-shrike	*Coracina novaehollandiae*					N	N	N	N
White-bellied Cuckoo-shrike	*Coracina papuensis*					N	N	N	N
White-winged Triller	*Lalage sueurii*					N	N	N	N
Olive-backed Oriole	*Oriolus sagittatus*					N	N	N	N
White-breasted Woodswallow	*Artamus leucorynchus*		√			SN	N	N	N
Masked Woodswallow	*Artamus personatus*		√			SN	N	N	N
White-browed Woodswallow	*Artamus superciliosus*		√			SN	N	N	N
Black-faced Woodswallow	*Artamus cinereus*		√			SN	N	N	N
Dusky Woodswallow	*Artamus cyanopterus*		√			SN	N	N	N
Grey Butcherbird	*Cracticus torquatus*					N	N	N	N
Pied Butcherbird	*Cracticus nigrogularis*					N	N	N	N
Australian Magpie	*Gymnorhina tibicen*			√		SN	SP	SN	SP
Pied Currawong	*Strepera graculina*			√		SN	SP	SN	SP
Grey Currawong	*Strepera versicolor*			√		SN	SP	SN	SP
Australian Raven	*Corvus coronoides*			√		SN	SP	SN	SP
Little Raven	*Corvus mellori*			√		SN	SP	SN	SP
White-winged Chough	*Corcorax melanorhamphos*			√		HN	SP	HN	SP
Singing Bushlark	*Mirafra javanica*					N	N	N	N
House Sparrow*	*Passer domesticus*			√		N	N	N	N
Zebra Finch	*Taeniopygia guttata*		√		√	SN	SP	N	SP
Double-barred Finch	*Taeniopygia bichenovii*		√		√	SN	SP	N	SP
Red-browed Finch	*Neochmia temporalis*		√	√	√	SN	SP	N	SP
Diamond Firetail	*Stagonopleura guttata*		√	√	√	SN	SP	N	SP

Mistletoebird	*Dicaeum hirundinaceum*					N	N	N	N
Welcome Swallow	*Hirundo neoxena*					N	N	N	N
Tree Martin	*Hirundo nigricans*					N	N	N	N
Fairy Martin	*Hirundo ariel*					N	N	N	N
Rufous Songlark	*Cincloramphus mathewsi*					N	N	N	N
Silvereye	*Zosterops lateralis*		√		√	SN	SP	SN	SP
Common Blackbird*	*Turdus merula*		√	√		SN	SP	SN	SP
Common Starling*	*Sturnus vulgaris*			√		SN	N	SN	N
Common Myna*	*Acridotheres tristis*			√		SN	N	SN	N

APPENDIX 4 - LIKELY EFFECT OF FIRE ON MAMMAL SPECIES

These data are the result of expert opinion from researchers at Arthur Rylah Institute, Monash University and Deakin University, and are best estimates of the likely response to low-intensity fire. Note that actual response to fire will vary substantially depending on factors such as timing, severity, extent, post-fire conditions and previous disturbance history. * introduced species, SN slightly negative, HP highly positive, SP slightly positive, N neutral, U unknown.

Name		Nesting/Sheltering		Foraging		Impact of fire in short or long-term			
Common	Scientific	Ground	Low (< 2 m)	Ground	Shrub layer	Spring burn, short-term	Spring burn, long-term	Autumn burn, short-term	Autumn burn, long-term
Short-beaked Echidna	*Tachyglossus aculeatus*	√		√		SN	SN	SN	SN
Yellow-footed Antechinus	*Antechinus flavipes*		√	√		SN	SN	SN	SN
Brush-tailed Phascogale	*Phascogale tapoatafa*					N	N	SN	SN
Fat-tailed Dunnart	*Sminthopsis crassicaudata*	√		√		SN	SN	SN	SN
Common Dunnart	*Sminthopsis murina*	√		√		SN	SN	SN	SN
Long-nosed Bandicoot	*Perameles nasuta*	√		√		SN	SN	SN	SN
Common Wombat	*Vombatus ursinus*	√		√		SN	SN	SN	SN
Koala	*Phascolarctos cinereus*					N	N	N	N
Common Brushtail Possum	*Trichosurus vulpecula*					N	N	N	N
Eastern Pygmy-possum	*Cercartetus nanus*					N	N	N	N
Sugar Glider	*Petaurus breviceps*					N	N	N	N
Squirrel Glider	*Petaurus norfolcensis*					N	N	N	N
Common Ringtail Possum	*Pseudocheirus peregrinus*					N	N	N	N
Feathertail Glider	*Acrobates pygmaeus*					N	N	N	N
Western Grey Kangaroo	*Macropus fuliginosus*	√		√		HP	HP	SP	SP
Eastern Grey Kangaroo	*Macropus giganteus*	√		√		HP	HP	SP	SP
Black Wallaby	*Wallabia bicolor*	√		√		HP	HP	SP	SP
Southern Freetail Bat (lp)	*Mormopterus planiceps*					N	N	N	N

White-striped Freetail Bat	*Tadarida australis*					N	N	N	N
Gould's Wattled Bat	*Chalinolobus gouldii*					N	N	N	N
Chocolate Wattled Bat	*Chalinolobus morio*					N	N	N	N
Lesser Long-eared Bat	*Nyctophilus geoffroyi*					N	N	N	N
Gould's Long-eared Bat	*Nyctophilus gouldi*					N	N	N	N
Inland Broad-nosed Bat	*Scotorepens balstoni*					N	N	N	N
Large Forest Bat	*Vespadelus darlingtoni*					N	N	N	N
Little Forest Bat	*Vespadelus vulturnus*					N	N	N	N
House Mouse*	*Mus musculus*	√		√		SP	SP	SP	SP
Black Rat*	*Rattus rattus*	√		√		SP	SP	SP	SP
Feral Dog*	*Canis familiaris*	√		√		U	U	U	U
Red Fox*	*Canis vulpes*	√		√		U	U	U	U
Cat*	*Felis catus*	√		√		U	U	U	U
Brown Hare*	*Lepus capensis*	√		√		HP	HP	SP	SP
European Rabbit*	*Oryctolagus cuniculus*	√		√		HP	HP	SP	SP

APPENDIX 5 - LIKELY EFFECT OF FIRE ON REPTILE AND FROG SPECIES

These data are the result of expert opinion from researchers at Arthur Rylah Institute, Monash University and Deakin University, and are best estimates of the likely response to low-intensity fire. Note that actual response to fire will vary substantially depending on factors such as timing, severity, extent, post-fire conditions and previous disturbance history. HN highly negative, SN slightly negative, N neutral, U unknown.

Name		Nesting/Sheltering		Foraging		Impact of fire in short or long-term			
Common	Scientific	Ground	Low (< 2 m)	Ground	Shrub layer	Spring burn, short-term	Spring burn, long-term	Autumn burn, short-term	Autumn burn, long-term
Common Long-necked Tortoise	*Chelodina longicollis*	√				SN	N	N	N
Tree Dragon	*Amphibolurus muricatus*	√		√	√	SN	N	SN	N
Eastern Bearded Dragon	*Pogona barbata*	√	√	√		SN	N	SN	N
Wood Gecko	*Diplodactylus vittatus*		√	√	√	SN	N	SN	N
Marbled Gecko	*Christinus marmoratus*		√		√	SN	N	SN	N
Southern Spiny-tailed Gecko	*Strophurus intermedius*		√	√	√	SN	N	SN	N
Thick-tailed Gecko	*Underwoodisaurus milii*	√		√		SN	N	SN	N
Pink-tailed Worm-Lizard	*Aprasia parapulchella*	√		√		SN	N	SN	N
Striped Legless Lizard	*Delma impar*	√		√		HN	SN	SN	U
Olive Legless Lizard	*Delma inornata*	√		√		HN	SN	SN	U
Burton's Snake-Lizard	*Lialis burtonis*	√		√		HN	SN	SN	U
Common Scaly-foot	*Pygopus lepidopodus*	√		√		HN	SN	SN	U
Hooded Scaly-foot	*Pygopus schraderi*	√		√		HN	SN	SN	U
Southern Rainbow Skink	*Carlia tetradactyla*	√		√		SN	N	SN	N
Carnaby's Wall Skink	*Cryptoblepharus carnabyi*	√	√	√		SN	N	SN	N
Large Striped Skink	*Ctenotus robustus*	√		√		SN	N	SN	N
Copper-tailed Skink	*Ctenotus taeniolatus*	√		√		SN	N	SN	N

Eastern Striped Skink	*Ctenotus orientalis*	√		√		SN	N	SN	N
Cunningham's Skink	*Egernia cunninghami*	√	√	√		SN	N	N	N
Black Rock Skink	*Egernia saxatilis intermedia*	√	√	√		SN	N	SN	N
Tree Skink	*Egernia striolata*		√	√		SN	N	N	N
White's Skink	*Egernia whitii*	√	√	√		SN	N	SN	N
Three-toed Skink	*Hemiergis decresiensis*	√		√		SN	SN	SN	N
Garden Skink	*Lampropholis guichenoti*	√		√		SN	N	SN	N
Bougainville's Skink	*Lerista bougainvillii*	√		√		SN	SN	SN	N
Grey's Skink	*Menetia greyii*	√		√		SN	SN	SN	N
Boulenger's Skink	*Morethia boulengeri*	√		√		SN	N	N	N
Blotched Blue-tongued Lizard	*Tiliqua nigrolutea*	√		√		SN	U	SN	U
Common Blue-tongued Lizard	*Tiliqua scincoides*	√		√		SN	U	SN	U
Stumpy-tailed Lizard	*Trachydosaurus rugosus*	√		√		SN	N	SN	N
Sand Goanna	*Varanus gouldii*	√		√		SN	U	U	N
Tree Goanna	*Varanus varius*	√	√	√	√	SN	U	U	N
Carpet Python	*Morelia spilota metcalfei*	√		√	√	SN	U	SN	U
Peter's Blind Snake	*Ramphotyphlops bituberculatus*	√		√		SN	U	SN	U
Gray's Blind Snake	*Ramphotyphlops nigrescens*	√		√		SN	U	SN	U
Woodland Blind Snake	*Ramphotyphlops proximus*	√		√		SN	U	SN	U
Eastern Tiger Snake	*Notechis scutatus*	√		√		U	N	SN	U
Red-bellied Black Snake	*Pseudechis porphyriacus*	√		√		SN	U	SN	N
Eastern Brown Snake	*Pseudonaja textilis*	√		√		SN	U	SN	N
Dwyer's Snake	*Suta dwyeri*	√		√		SN	U	SN	N
Little Whip Snake	*Suta flagellum*	√		√		SN	U	SN	N
Mitchell's Short-tailed Snake	*Suta nigriceps*	√		√		SN	U	SN	N
Bandy Bandy	*Vermicella annulata*	√		√		SN	U	SN	N
Plains Froglet	*Crinia parinsignifera*	√		√		SN	U	U	N

Common Froglet	*Crinia signifera*	√		√		SN	U	U	N
Sloane's Froglet	*Crinia sloanei*	√		√		SN	U	U	N
Southern Bullfrog	*Limnodynastes dumerilii*	√		√		SN	U	U	N
Spotted Marsh Frog	*Limnodynastes tasmaniensis*	√		√		SN	U	U	N
Barking Marsh Frog	*Limnodynastes fletcheri*	√		√		SN	U	U	N
Mallee Spadefoot Toad	*Neobatrachus pictus*	√		√		SN	U	U	N
Common Spadefoot Toad	*Neobatrachus sudelli*	√		√		SN	U	U	N
Bibron's Toadlet	*Pseudophryne bibronii*	√		√		SN	U	HN	SN
Southern Toadlet	*Pseudophryne semimarmorata*	√		√		SN	U	HN	SN
Southern Brown Tree Frog	*Litoria ewingii*	√	√	√	√	SN	U	U	N
Plains Brown Tree Frog	*Litoria paraewingi*	√	√	√	√	SN	U	U	N
Peron's Tree Frog	*Litoria peronii*	√	√	√	√	SN	U	U	N
Growling Grass Frog	*Litoria raniformis*	√	√	√	√	SN	U	U	N