

Catering for the needs of fauna in fire management: science or just wishful thinking?

Michael F. Clarke

*Department of Zoology,
La Trobe University,
Bundoora, 3086, Australia
m.clarke@latrobe.edu.au*

10 Key words: fire ecology, fauna, vital attributes
Running title: Fire – catering for the needs of fauna
Viewpoint Article for *Wildlife Research*

Abstract

Ecological fire management in Australia is often built on an assumption that meeting the needs of plant species will automatically meet the needs of animal species. However, the scarcity of ecological data on the needs of fauna in relation to fire undermines the confidence managers should place in current popular frameworks for planning ecological burning. Such frameworks are built almost entirely around the goal of maintaining plant community diversity. They provide little guidance to managers regarding the characteristics of desirable ‘mosaics’ (e.g. patch size, connectivity or composition of age-since-burnt classes) or the timing of fires in relation to faunal population trends linked to other cycles (e.g. El Nino events). Claims by agencies of adopting an adaptive management approach (‘learning by doing’) to cope with a dearth of knowledge are credible only if monitoring and evaluation are carried out and future actions are modified in light of new evidence. Much monitoring of fauna is of such a small scale and short duration that the statistical likelihood of detecting a positive or negative effect of the management regime is minute. Such shortcomings will only be overcome through broad-scale and/or long-term studies of fauna. The funding for such research is unlikely to be forthcoming if fire ecologists and land managers convey the impression that the current data are adequate for the implementation of the current planning frameworks.

Introduction

Recent large fires in Australia (e.g. the Alps 2003, 2005, Central Kimberley 2006, the Goldfields region of Western Australia 2008) have put the spotlight on land management agencies and their fire management policies and practices. Irrespective of whether the fire in question began as a lightning strike or a prescribed burn that escaped control lines, often people’s views have tended to polarise fairly quickly into two camps. In one camp are those who are quick to pronounce that the fire is an *ecological*

disaster that could, and should, have been avoided had the land management agency done *x*, *y* and *z*. Just as quickly others, often from agencies or governments, reassure the public not to worry, that ‘*fire is a natural component of Australian ecosystems*’ and that soon they will witness the wondrous regeneration of the bush back to its former glory. This polarisation of views unhelpfully masks how little is known about the needs of fauna, when it comes to fire, or the best ways to meet those needs. I wish to argue that, when it comes to understanding the long-term impact of fire upon fauna, in many cases there is insufficient ecological evidence to confidently side with either camp.

At least two major challenges confront ecologists and land managers in regard to fire ecology. First, what should we be aiming for? What is the range of ecologically acceptable outcomes that will ensure the long-term survival of all indigenous taxa in the landscape and how does one determine desirable and undesirable fire regimes for different communities? The second challenge, that is not the focus of this paper but of equal importance, is how can managers achieve the desired outcomes in regard to fire management given the highly modified and fragmented nature of the habitats being managed. While both of these challenges need urgent attention, if we do not know what we are aiming for at a landscape scale in fire management, in terms of spatial and temporal heterogeneity, then we will never be in a position to critically assess whether our methods of fire management have been successful. Consequently, the focus of this paper is an examination of current approaches being used in Australia to identify and define desirable ecological outcomes in regard to fire.

In most states in Australia management agencies have statutory obligations to protect human life and property from wildfire, and also to conserve the richness and viability of species and ecological communities within a region while maintaining ecological processes through their management of fire (e.g. Department of Sustainability and Environment 2006). Management agencies attempt to achieve these two broad objectives through the prevention and suppression of wildfires and the application of prescribed burning (where fire is applied under specified environmental conditions to a predetermined area and at the time, intensity of heat and rate of spread required (e.g. Fire Ecology Working Group 2004)).

In this article I examine how well four common premises that appear to underpin current fire management practices in Australia encompass the needs of fauna. They are:

1. that pyrodiversity begets biodiversity (landscapes exposed to a greater diversity of fire regimes equate to landscapes with greater biodiversity (Parr and Andersen 2006)).
2. that organisms are more likely to be able to cope with disturbance regimes with which they are evolutionarily familiar;
3. that our current knowledge of the needs of plants in regards to fire is a sufficient basis upon which to determine over and under-represented seral stages of a vegetation type in a landscape;
4. that burning one over-represented age class of a vegetation type reliably produces the desired under-represented age class of that same vegetation type.

My aim is to consider if these premises form a sound basis for conserving fauna, as well as flora. While there have been some important detailed studies of the response to fire of select animal species or groups (e.g. splendid fairy-wren *Malurus splendens* Brooker 1998; ground parrot *Pezoporus wallicus wallicus* Meredith *et al.* 1984, Baker and Whelan 1994; small mammals in forest Christensen and Kimber 1975, heathland, Fox 1982, 1990, Cockburn *et al.* 1981 or hummock grassland, Letnic and Dickman 2005), the general scarcity of data for fauna, as compared to flora, has been highlighted by several authors (e.g. Bradstock and Cohn 2002). They stressed that in identifying the needs of fauna in fire management ‘...the critical proportion of a particular seral state in a landscape is so far undefined for any animal species’ and that ‘... the appropriate degree of spatial heterogeneity for animal species has received virtually no attention [my emphasis]’. Some significant progress, particularly in northern and central Australia, has been achieved in recent years in gaining at least a better qualitative understanding of the needs of some fauna in regards to fire (e.g. Letnic and Dickman 2005; Letnic *et al.* 2005, Woinarski *et al.* 2005). Nevertheless, there remains an urgent need for ecologists to address the significant gaps in our knowledge of the needs of fauna in regards to fire. With the exception of a limited number of threatened animals (e.g. heath mouse *Pseudomys shortridgei* Wouters 1993; western bristlebird *Dasyornis longirostris*, noisy scrub-bird *Atrichornis clamosus*, western ground parrot *Pezoporus wallicus flaviventris* Burbidge 2003; Gilbert’s potoroo *Potorous gilbertii* Courtenay and Friend 2004), ecological fire management currently attempts to maintain the diversity of *plant* communities within a reserve or landscape and assumes that this will also maintain faunal diversity. Policy makers and land managers need to engage in a more rigorous consideration of the needs of fauna in regards to fire and to move beyond such broad assumptions and wishful thinking.

100 **The ecological basis of current approaches to fire management**

The four fundamental ecological premises that appear to underpin current approaches to fire management in Australia all seem ecologically reasonable. However, all suffer from such profound knowledge gaps that they provide little concrete direction for land managers attempting to make detailed plans for the conservation of flora and fauna in a particular reserve or across broad landscapes.

Premise 1: Pyrodiversity begets biodiversity

Many managers assume that landscapes exposed to a greater diversity of fire regimes (‘pyrodiversity’) equate to landscapes with greater biodiversity (Parr and Andersen 2006). Traditionally the term *fire regime* encompasses components such as time-since-last-burn, fire frequency, duration of inter-fire interval, fire intensity and seasonal timing (Gill 1981). It is worth noting that the spatial components of fire regimes that are potentially of more importance to animals than plants (e.g. patch size, proximity to similar habitats, connectivity between patches) have often been missing from such definitions (e.g. Gill 1975, 1981, 1999, Department of Sustainability and Environment 2006). As several authors have stressed (e.g. Friend 1993, Whelan *et al.* 2002, Kenny *et al.* 2004), our current understanding of the relative

importance of any components of the fire regime for the persistence of *fauna* within a landscape is extremely limited. Some would argue it is little better for flora (Cheal 2004).

The focus on pyrodiversity has arisen from numerous studies finding different assemblages of organisms occur at sites at different seral stages in the successional process following a fire (e.g. Woinarksi et al. 1999). Authors of such studies often then conclude their paper with a recommendation that a policy of mosaic burning will maximise species diversity (e.g. Masters 1993). However, such studies have rarely ever been designed to sample the fauna at the scale of the mosaic and are consequently unable to provide any guidance as to the minimum patch size, or proximity and connectivity to similar or different patches within the fire mosaics. Nor do they present any evidence that the management practice they are advocating (unspecified mosaic burning) will actually maximise species diversity or inhibit large-scale fires. Nevertheless, in light of such recommendations, the aim of many fire plans is to avoid uniformity in fire history over large areas; in particular avoiding fires that are large relative to the size of a reserve (e.g. Willson 1999) through some level of active management (e.g. fire prevention, suppression or ecological burning). However, beyond this very basic recommendation that uniformity in fire history should be avoided lies a much more challenging question (Bradstock *et al.* 2005): Which mosaic should managers be aiming for? Are all equally desirable? Will all assist in maintaining the diversity of animals, as well as plants? There is a dearth of studies that have actually compared floral or faunal diversity across coarse or fine-grained mosaics, as opposed to just individual sites of differing fire history. Consequently, with the exception of total uniformity in fire history across a landscape, there is currently an absence of measurable criteria by which one can identify desirable or undesirable mosaics. Typically, the assumption is made that pyrodiversity begets diversity in plant communities and plant community diversity begets faunal diversity.

Testing this assumption requires detailed analysis of the response of faunal and floral groups to differing fire regimes at a variety of spatial and temporal scales. However, because of a perception that plants are more easily surveyed than animals, a select group of plants are typically used as surrogates for faunal diversity (e.g. Fire Ecology Working Group 2002, Kenny et al. 2004). Generally the suitability of these indicator species to act as surrogates for the diversity and abundance of faunal groups is assumed, rather than tested (Noss 1990, Keith *et al.* 2002).

There are several reasons why we might expect animals to respond differently to fire than plants, and therefore for plant diversity to not always be a good surrogate for faunal diversity. Plants have numerous well-documented strategies for persisting at a site and recovering after fire on that same site, such as hard woody protection for seeds, re-sprouting from tough ligno-tubers, thick bark protecting epicormic buds, resilient seeds that survive for decades in the soil seed bank etc (Good 1981). Although more mobile animals may be able to avoid incineration by the fire, increased predation pressure due to lack of shelter, or a shortage of food post-fire may increase mortality rates (Noble 1982). Less mobile animals may become temporarily extinct at a site following a fire and rely on re-colonisation of the site from another

source (e.g. slender-billed thornbill *Acanthiza iredalei* Ward and Paton 2004). Provided there are sufficient source populations within dispersing distance, and the key habitat features used by the animal recover at the burnt habitat, then the animals may well re-colonise. But if the burnt site is isolated or the key features do not recover before the next fire event, then the animals may become locally extinct at the site, due to a lack of re-colonisation (Whelan 1995).

160 Furthermore, in contrast to plants, some animals need to range over a number of habitat types with different fire history to meet their daily or annual resource requirements (Keith *et al.* 2002). For example, Woinarski (2005) reported that masked finches *Poephila personata* and partridge pigeons *Geophaps smithii* require both burnt and unburnt areas within their home range (1-2 ha) if they are to have nest sites secure from predation, as well as sufficient seed to sustain breeding. If all the key resources these species require are not in close proximity, they will struggle to persist in the landscape. Other species like the black-eared miner *Manorina melanotis* only breed in very large patches of long unburnt mallee. Individuals belonging to a single colony may range over 4-8 km² and the minimum area of old-growth mallee needed to sustain a viable population is likely to be around 32 km² (M. Clarke, unpubl. data). The spatial configuration of some mosaics that may sustain plant diversity, may prove inadequate to
 170 sustain viable populations of animals. As a growing number of authors have stressed, gathering empirical data to identify a range of desirable and undesirable mosaics is an urgent task in fire ecology around the world, particularly as remaining habitats become increasingly fragmented (e.g. Bradstock *et al.* 2005; Parr and Andersen 2006). Without such data, recommendations that managers work to maximise pyrodiversity or aim for an undefined 'mosaic' of fire histories lack precision and can be used as the ecological justification for almost any fire management action.

Premise 2: Organisms are more likely to cope with disturbance regimes with which they are evolutionarily familiar

180 During their evolutionary development, flora and fauna have experienced particular disturbance regimes due to fire. Some like grasslands or heathlands may have burnt often; others, like wet sclerophyll forests, may have experienced a canopy-killing fire only once every couple of centuries (Woinarski 1999). These different spatial and temporal patterns of disturbance are likely to have acted as profound selective forces resulting in organisms with life-history attributes that enable them to cope with a particular range of disturbances, e.g. fire regimes, floods and droughts (Noble and Slatyer 1980).

The desirability of attempting to replicate past disturbance regimes such as those generated by traditional Aboriginal burning practices remains a contentious issue in many parts of the country (e.g. Benson and Redpath 1997; Jurskis 2005; Abbott 2003); in part due to varying confidence as to the scale, frequency, timing or intensity of the Aboriginal burning practices one might attempt to simulate in particular
 190 habitats. In some parts of the country attempts have been made to assemble historic records regarding the precise nature of traditional Aboriginal burning practices (e.g. Abbott 2003). However, more often management plans invoke unspecified Aboriginal burning practices when justifying the need for fire in

the landscape (e.g. Department of Sustainability and Environment 2007). Several authors have argued against attempting to return landscapes to pre-European fire regimes (e.g. Keith and Henderson 2002), given the additional environmental pressures many habitats now face. For example, there may be very good reasons for trying to avoid fires of historic proportions if they reduce entire reserves to a single seral stage. Therefore, when agencies reassure the public that ‘fire is a *natural* component of Australian ecosystems’, how the word ‘natural’ is defined really matters. A problem arises if, through either extinguishing all fires or becoming over zealous in prescribed burning, a novel fire regime is imposed upon a community of organisms, with which it is ill-equipped to cope. The organisms occupying that habitat may never have had to recover from fires of that frequency, size, intensity or at that time of the year. Furthermore, since many conservation reserves are effectively islands in a sea of agriculture, the modern landscape-context has become critically important. Large fires, that in the past were of a magnitude that posed little threat to a species’ survival, may now be catastrophic due to the isolating effect of habitat fragmentation or loss of key resources due to grazing pressure from domestic or feral stock (Woinarski et al. 2005).

Nevertheless, the premise that an organism’s capacity to regenerate or re-colonise following fire will be limited by the organism’s own life-history attributes (e.g. age at sexual maturity, reproductive rate, dispersal ability) seems inescapable. Therefore, much effort has been made, to identify what Noble and Slatyer (1980) labelled the ‘Vital Attributes’ of organisms, largely plants. Knowledge of how long it takes before a plant produces viable quantities of seed to replace itself or how long a seed that requires fire to germinate remains viable in the soil seed bank, has been used as a basis for calculating minimum and maximum tolerable fire intervals for a wide range of vegetation types in Australia (e.g. Fire Ecology Working Group 2002, Kenny *et al.* 2004). Using the persistence of particular fire sensitive plant species in a community (‘Key Fire Response Species’, Tolhurst and Friend 2001) as indicators of ecosystem health shares many similarities with the concept of the ‘umbrella species’; whose requirements are believed to encapsulate the needs of other species. The umbrella species concept has been further refined by Lambeck (1997) in his ‘focal species approach’, whereby the species whose populations are most limited by a particular process, such as fire, is identified, and thereafter used to define the levels at which fire must be managed for the good of the rest of the members of that ecological community.

Unfortunately, at least three factors currently limit the power of this approach. First, it is much more straightforward to gain vital attribute data for plants than it is for animals. Plants are sessile for most of their life cycle, whereas animals move, making the monitoring of the latter’s development and reproductive performance more difficult. Consequently, vital attribute data bases are almost exclusively based on plants, with few or no data for animals (e.g. Fire Ecology Working Group 2002, Kenny et al 2004). So again, the assumption is made that the minimum and maximum tolerable fire intervals that we postulate will meet the needs of a select group of plants, will also meet the needs of the fauna in the landscape. The confidence that some place in this assumption is well-illustrated by a quote from Kenny

et al. (2004, p.36) in the *Guidelines for Ecologically Sustainable Fire Management* for New South Wales:

‘The proposed fire intervals [for wet sclerophyll forests of 25-60 years], derived from floristic analysis, are compatible with the requirements of threatened fauna with known fire response information.’

This sounds reassuring until one turns to the very next page of the report which states that of the 55 species of threatened fauna found in this vegetation type, ‘good’ fire response information is available for only two species (let alone the other animals that have yet to decline sufficiently to make it on to the state’s threatened species list)! While the authors frankly concede in the project summary that ‘Data on threatened species of fauna were generally inadequate’, they nevertheless go on to conclude that ‘the available data indicated that they [threatened fauna species] were unlikely to be disadvantaged by fire regimes within the domain of guidelines specified for relevant vegetation formations.’ Such confidence that floristic diversity is an adequate surrogate for faunal diversity is not confined to New South Wales (e.g. Fire Ecology Working Group 2002).

Furthermore, current policy documents making recommendations regarding appropriate fire intervals do not indicate what proportion of species in a vegetation community need to be represented in a Vital Attributes data set for it to be deemed an adequate basis upon which to calculate minimum and maximum tolerable fire intervals. So sparse have the vital attributes data been in the past, some studies have pooled species data from a range of related, but inadequately sampled plant communities in order to generate fire intervals for even broader habitat types (Fire Ecology Working Group 2002). This may make analysis numerically possible in the absence of data for certain communities, but whether the recommendations arising from this application of Noble and Slatyer’s (1980) method have any ecological validity is debateable. Ecologists know the predictive ability of a model will always be constrained by the quality and quantity of the data entered into the model (‘garbage in – garbage out’), and the assumptions upon which the model is built (Quinn and Keogh 2002). Alarm bells should begin to ring when maximum tolerable fire intervals of 50 years are generated for vegetation types that have *never* been recorded burning (Fire Ecology Working Group 2002), like Saline Shrubland (Cheal 2004). So even if one accepted the assumption that floristic diversity will reliably equate to faunal diversity, it is questionable whether current vital attributes data are sufficient to form an adequate basis for estimating minimum and maximum tolerable fire intervals, even for plants.

Second, data on the length of time that seeds remain viable within a soil seed bank is often lacking in the vital attributes data-base for many species (Cheal 2004), despite Noble and Slatyer (1980) recommending that soil seed-bank longevity data were an essential component of the *minimal* data set needed to apply their method. The absence of such data increases the likelihood of underestimating how resilient the seed bank is for a species and may result in an ‘apparently’ precautionary approach of burning a site sooner

270 rather than later if the plant is no longer visible above ground (i.e. out of fear that it has become locally extinct). The case of wheel-fruit (*Gyrostemon australasicus*) in Wyperfeld National Park, Victoria, provides a sobering example that highlights the remarkable resilience of some seeds. This plant was extremely rare within Wyperfeld National Park east of Outlet Creek between 1915 and 1984, but became one of the most abundant plants in some parts of the park for about five years following a fire in 1984. Although not conspicuous above ground, it had been present in the park for the 69 years between fires in the soil seed-bank (D. Cheal, pers. comm.).

280 Third, the structural attributes of a habitat (e.g. hollows in which to nest) that are clearly essential for reproduction by a wide range of fauna may not arise until decades after the Key Fire Response Species is capable of producing viable quantities of seed. It is often primarily the *structural features* of a plant species that are of importance to fauna, not just the plant's *presence* at a site (e.g. Caughley 1985; Schlesinger *et al.* 1997; Mackey *et al.* 2002). For example, many arboreal mammals or forest owls require large hollows as breeding sites (Gibbons and Lindenmeyer 1997; Mackey *et al.* 2002), but such hollows develop slowly, particularly in Australian hardwoods (e.g. 144-194 years in blackbutt *Eucalyptus pilularis*, Mackowski (1984)). Current prescriptions of inter-fire intervals of 25-60 years wet sclerophyll forests in New South Wales (Kenny *et al.* 2004), based on vital attributes data for Key Fire Response Species of plants, are unlikely to deliver the hollows these animals need to breed.

290 A significant limitation in interpreting and implementing these kinds of prescriptions is that they rarely are explicit in regards to the *intensity* of the fire being prescribed at the recommended inter-fire interval (e.g. Fire Ecology Working Group 2002); does the inter-fire interval refer to a canopy-destroying fire or just to a fire that removes the understorey? In some habitats low intensity fire may stimulate the regeneration of understorey species but not kill canopy trees, allowing the trees to continue aging and develop hollows in due course. In the case of Kenny *et al.* (2004), they do suggest that in NSW wet sclerophyll forests 'crown fires should be avoided in the lower end of the range' (25-60 years). Small-scale crown fires in 60 year-old wet sclerophyll forest may indeed create additional breeding habitat for hollow-dependent fauna; since in some circumstances high intensity fires can enhance the development of hollows, as well as destroy them. However, the climatic conditions necessary to achieve a crown fire in such habitats might make constraining such a fire to a small area problematic.

300 A fire that does not kill the canopy species at a site may nevertheless still have a profound and long-term impact upon a range of different fauna. For example, many animals are dependent upon thick litter layers accumulating on the ground; species as diverse as the burrowing frog *Myobatrachus gouldii* (Bamford 1992), the malleefowl *Leipoa ocellata* (Benshemesh 1990), and numerous invertebrates (York 1999). Unfortunately, the depth of litter that an animal might require as prime breeding habitat, might be regarded as the critical depth of fuel triggering the need for fuel reduction burning (e.g. McCarthy *et al.* 1999). If managers consistently maintain fuel loads on the forest floor at levels that will inhibit the spread of fires, species that require deep litter layers may be put at risk (e.g. malleefowl, Benshemesh 1990).

310 Canopy scorching as a result of low intensity fire may significantly reduce the availability of key food resources for some species, such as seeds and nectar. For example, the endangered south-eastern red-tailed black-cockatoo *Calyptorhynchus banksii graptogyne* is a dietary specialist that feeds predominantly upon the seeds of brown stringybark *Eucalyptus baxteri* and desert stringybark *Eucalyptus arenacea* (Koch 2005). The heathy understorey of the forests it occupies has been burned to maintain floristic diversity of the heaths. However, fires that burn or scorch the canopy result in reduced seed production for up to nine years (Koch 2005). So although neither the floristic composition nor the age-class structure of the canopy vegetation may have changed at a site, ecological burning has the potential to change the site in a way that diminishes its value to the endangered red-tailed black-cockatoo. Similarly, apiarists working in Victorian mallee vegetation report significant reductions in eucalypt flowering and nectar
320 production for many years following fires (Eric Whyte, pers. comm.). Given the typically large scale of fires in mallee vegetation (e.g. 180,000 ha Taylorville fire in 2006), this must represent a significant and prolonged loss or reduction of key resources for nectarivorous birds, mammals and insects in that habitat.

Knowledge that a Key Fire Response Species has been sufficiently resilient to persist at a site following a fire is clearly not enough for ecological management of the entire community. Has enough time elapsed with suitable climatic conditions for vegetation at the site to develop the key habitat attributes like a hollow, a deep litter layer, a seed crop or a substantial nectar flow? Furthermore, have these attributes persisted on the site long enough for the animal population to recover sufficiently to perpetuate itself? So although plant diversity may not be irreversibly altered at a site by a single fire or series of fires, other
330 effects of ecological burning can be quite subtle and potentially have profound consequences for other components of the community.

Given the acknowledged shortage of data available on the responses of fauna to fire (e.g. Kenny *et al.* 2004), it is particularly puzzling why the data on key habitat variables that are available are not applied more widely in the calculation of appropriate inter-fire intervals. It is odd that prescribed burning of a forest type can be postponed until a Key Fire Response Plant Species has set sufficient seed to perpetuate itself, and yet there appears to be a reluctance to apply a similar level of patience when it comes to ensuring that the structural features are in place that would enable the animal to do the same. It is encouraging that some agencies have recently begun reviewing their recommended inter-fire intervals in
340 the light of the requirements of fauna and have begun to include explicit recommendations in regard to the intensity of fire being prescribed (D. Cheal, unpubl. data).

For most animal species we have yet to identify the structural features of the vegetation (e.g. depth of litter, size of hollows, abundance of seed) that provided key resources for survival and reproduction. If we are to move beyond simplistic assumptions that plant community diversity will deliver animal diversity, we need to couple identification of key structural features with a deep understanding of the processes that influence their development. Some excellent studies in northern Australia are starting to

shed light on the complexity of the spatial and temporal combination of factors (e.g. fire, rainfall and grazing pressure) required to ensure the essential resources needed by some species are present in the landscape (e.g. production of seed eaten by Gouldian finches *Erythrura gouldiae*, Dostine et al. 2001, Woinarski et al. 2005). Such studies illustrate the value of autecological studies and highlight that the development of key resources or structural features are likely to require a combination of events and interactions between abiotic (e.g. fires, floods, droughts, cyclones) and biotic components of the environment (e.g. grazing pressure, plant pathogens and predators), rather than just a single fire event (Gibbons and Lindenmeyer 1997). Once an understanding is gained of the processes that influence the development of the key structural features of the vegetation, the next challenge is to understand the impact of anthropogenic changes upon those processes, and the capacity of the organisms to cope with such changes (Woinarski et al. 2005)

360 *Premise 3: that our current knowledge of the needs of plants in regards to fire is a sufficient basis upon which to determine over and under-represented seral stages of a vegetation type in a landscape*

Some agencies are using the minimum and maximum tolerable fire intervals derived from the vital attributes of a select group of key fire response species (all plants) to generate an ideal (desirable) age-class distribution for each particular vegetation community. Their aim is to determine the desirable proportion of a vegetation type to have in each time-since-last-burnt age-class. The approach (e.g. Tolhurst and Friend 2001) has been to insert the estimates of minimum and maximum tolerable fire intervals (derived from the vital attributes of a select group of plants) into a single mathematical model (based on a homogenous negative exponential distribution) that will generate ‘ideal’ age-class distributions for that vegetation type. The use of this particular distribution assumes the time since last

370 burnt does not influence the probability of ignition (flammability) (Johnson and Gutsell 1994). While this assumption may hold for some vegetation types, it seems unrealistic for many in which the probability of ignition is likely to change along with changes to the litter layer (Good 1996; Tolhurst and Friend 2001; Mackey *et al.* 2002). Furthermore, it is exceedingly unlikely that all vegetation types will share a *common pattern* to their probability of ignition over time since last burnt. Bradstock *et al.* (2005, p.241) cautioned against ‘predicating intervention solely on the basis of time-since-fire distributions, particularly where the intent is to stabilise or manipulate such distributions to conform to some ideal.’. Nevertheless, the same mathematical model was applied to 157 vegetation types in Victoria (Fire Ecology Working Group 2002). Gill and McCarthy (1998), while advocating a negative exponential distribution, cautioned that ‘There appears to be no single type of probability-distribution function to apply universally to

380 intervals between fires at any one point of a landscape.’. Similarly, Williams *et al.* (1994) stressed that fire prescriptions were not necessarily ‘portable’ from one site to another, because the responses to fire will vary between sites and plant communities. Nevertheless, the Fire Ecology Working Group (2002) claimed that irrespective of variance in flammability, they anticipated ‘relatively high proportions of young age-classes’ and ‘relatively small areas of very old-age classes’ when defining the ‘idealised’ distribution of age-classes for *all* Victorian vegetation types. Cool temperate rainforests, that include genera like *Nothofagus*, require long periods free of fire to become established (Busby 1986), whereas

many heathland species do not (Gill 1999). It is hard to envisage ‘relatively high proportions of young age-classes’ and ‘relatively small areas of very old-age classes’ being the ecologically sound age-class distribution pattern across *all* vegetation types in any state. Furthermore, the approach seems to ignore
 390 the possibility of one vegetation type (e.g. wet sclerophyll forest) under-going a state shift into a different vegetation type (e.g. cool temperate rainforest) in very low or very high frequency of burning.

Premise 4 That burning one over-represented age class of a vegetation type reliably produces the desired under-represented age class of that same vegetation type

The final premise that most ecological burning in Australia appears to be built upon is the assumption that by burning one seral stage of a vegetation type the desired younger seral stage of the same vegetation type can be created, irrespective of the timing of the fire. This assumption is rarely tested by detailed pre and post-fire monitoring. The poor regeneration by coast banksia *Banksia integrifolia* following the 2005 fire at Wilsons Promontory, Victoria (Neild 2006) or *B. ornata* following severe frosts prior to burning in
 400 Wyperfeld National Park (O’Brien 1989) should provide sobering reminders that burning one seral stage does not guarantee the creation of the younger seral stage of the same vegetation type. Other drivers, in addition to fire (e.g. adequate rainfall) are likely to be required to reliably generate earlier seral stages of a vegetation type.

Some vegetation communities and animal populations may have evolved to cope with fires that occur at particular points in the annual or El Niño rainfall cycle, but not others. For example, Griffith and Christian (1996) reported that frillneck lizard *Chlamydosaurus kingii* benefited from less intense early dry season fires because risk of incineration was low and open patches were created in the habitat for foraging. By contrast, late dry season burns resulted in much higher mortality levels. Large fires in
 410 mallee habitats are typically associated with above-average antecedent rainfall (Noble and Vines 1993). If such periods are associated with an abundance of resources, we might expect animal populations to also be peaking at such times. They would therefore be in a more robust position for recovery after the fire, as opposed to a fire coming at the end of years of drought.

Unfortunately, with many major wildfires in recent years there is increased public pressure on politicians and agencies for more extensive prescribed burning to protect assets, particularly during droughts when flammability levels are high. Such pressure creates an urgency for burning simply to ‘meet the targets’ or so as to be ‘seen to be doing something’, irrespective of whether or not such burning will actually protect assets or achieve ecologically desirable outcomes (Fernandes and Botelho 2003, Cary et al. in review).
 420 Such pressure promotes agencies seizing every window of opportunity to carry out prescribed burning on those rare days when it is safe to do so. Little consideration appears to be given as to whether animal populations in the region to be burnt have the numerical capacity to recover from the prescribed burn, e.g. are population levels currently so low that a prescribed burn may cause local extinction of the species. Without pre-burn monitoring of at least some animal populations, we may never know, until it is too late.

Unrealistic expectations of management agencies

The very label ‘management agency’ may foster an unhelpful expectation in the public’s mind that agencies should always be capable of ‘managing’ fire. When agencies apparently ‘fail’ to do so, 430 spokespersons and agency media units feel compelled to reassure us that no irreparable harm has been done, since our flora and fauna are ‘used to coping with fire’. Interestingly, no such reassurances are seen as necessary after other natural cataclysmic events like cyclones; presumably because the public has no expectation that land management agencies could have done anything to prevent such events, and so are ‘off the hook’. To achieve the level of fire control some in the media are calling for would require prescribed burning on a scale and frequency such that ground level fuels would be continually removed from large areas (Morrison et al. 1996); an ecologically undesirable outcome. While it is reasonable to expect management agencies to have taken reasonable preparatory measures to limit the negative effects of large fires, it needs to be acknowledged that the capacity of management agencies to control wide- 440 spread wild fires ignited by multiple lightning strikes in drought conditions on days of extreme fire danger is going to be similar to their capacity to control cyclones. If the public could allow agencies and their political masters to become a little less defensive, the climate might be right for a more rigorous examination and refinement of the ecological basis of fire management in this country.

Possible steps forward

In conclusion, I think there are seven steps that need to be taken to identify and define desirable ecological outcomes in regard to fauna and fire. First, there is an urgent need to acknowledge how inadequate our current data bases and understandings are, as a basis for determining desirable fire regimes for different vegetation types. Policy documents should acknowledge, rather than gloss over, these major gaps in our understanding of the role of fire in Australian ecosystems (e.g. Fire Ecology 450 Working Group 2004; Kenny *et al.* 2004). In this age of global warming, governments and the public need to be engaged in a more sophisticated discussion about the complexities of coping with fire in Australian landscapes. If politicians or senior bureaucrats are told that the current framework is working well, they can hardly be criticised for not funding the research that is needed to improve our understanding.

Second, *all* components of the adaptive management cycle (‘learning by doing’), advocated in most fire management plans, need to be adequately resourced and implemented. Adaptive management involves a cycle with five basic components: planning, implementation, monitoring, evaluation and finally adjustment and refinement of actions. Given the public pressure to protect assets and the Australian 460 admiration for ‘doers’, it is not surprising that in this country we have been very big on the first two components of the adaptive management process and extremely weak on the remaining three (Burgman and Lindenmeyer 1998; Williams *et al.* 2003). It is encouraging that in recent years the necessity of ongoing monitoring is being stipulated in some state Codes of Practice. For example, the *Code of Practice for Fire Management on Public Land* in Victoria (Department of Sustainability and

Environment 2006, p. 21, clause 237) states that the Department ‘must conduct or arrange soundly based sampling on an ongoing basis to monitor the effects of burning on fuel levels, flora, fauna and other values as appropriate’; in particular, species or vegetation types likely to be sensitive to prescribed burning regimes. Without effective monitoring of the long-term effects of management actions upon flora and fauna, claims of genuine adaptive fire management have a very hollow ring. We will also never improve upon our current frameworks and, in the worst-case scenario, may do irreparable ecological damage. We need to admit how little genuine ecological adaptive management is actually occurring; and then ensure our propensity for burning is matched with an equivalent fervour for learning.

Third, future consideration of the impact of fire regimes needs to move beyond floristics and take a multidisciplinary approach that examines the responses of a much wider range of taxa (e.g. fungi, invertebrates, reptiles, amphibians, birds and mammals, Woinarski 1999), as well as physical parameters (e.g. nutrient and resource availability). Until recently, some states have not even routinely mapped the extent of all fires within their jurisdiction, let alone monitor the responses of flora and fauna to fire.

Fourth, there needs to be a commitment to long-term monitoring (Gill et al. 2002). Too often in the past there have been short-term bursts of frenetic monitoring activity following major fire events. Ecologists are guilty of rushing to exploit the all-to-rare, but regrettably short-lived funding that appears as agencies and governments attempt to be seen to be doing something post-fire. Consequently, only obvious changes that occur in the first few years after the wildfire are likely to be detected. The need for ecologists to generate publishable results in their lifetime (preferably before the next grant application is due) may have contributed to a preponderance of studies in plant communities that exhibit rapid responses to fire. For example, although heathland covers less than 1% of the Australian mainland (Department of the Environment and Water Resources 2007), 46% of botanical studies reviewed by Gill (1999) were conducted in heathlands. The rapid responses to fire of such communities are not necessarily representative of most Australian plant communities, that may take decades or centuries to proceed through a complete progression of seral stages (Gill 1999) and whose outcomes may be influenced by other ecological events like frosts, droughts and cyclones (Chandler 1973, Letnic *et al.* 2005). And yet much of the current thinking about the role of fire in Australian landscapes is based on studies in coastal heathlands, because these are the places that have been most studied. Short-term studies will not shed much light on the long-term impacts of fires upon plant and animal communities whose successional processes may run over decades or centuries. If similar or greater effort could be spread over several decades with targeted monitoring every five to ten years then more profound insights might be gained (e.g. Edwards et al. 2003, Ward and Paton 2004, Woinarski et al. 2004). Such studies will become increasingly valuable the longer they run, but will need to run for decades.

Fifth, a commitment to long-term monitoring needs to be accompanied by targeted strategic experimental studies. The range of fire prevention and suppression methods being applied throughout the country and the stated commitment of most state agencies to an adaptive management approach should create numerous opportunities for experimental studies of the impact of alternative approaches upon fauna and

flora. Indeed some fire ecologists argue that a controlled experimental approach, based around purposefully-lit prescribed burns, leads to the greatest insights into the processes influencing recovery by fauna after fire (e.g. Friend 1996). However, the limitations, as well as the strengths of experimental studies should lead us to be cautious in applying the findings of such studies more broadly. The difficulty of generating true replicates in experimental burning is profound given local variations in land-form, fuel loads and changes in climatic conditions preceding, during and following a burn (e.g. did all the 'replicates' receive similar climatic conditions post-burning? Letnic *et al.* 2005). Such climatic factors introduce 'noise' into the data that greatly diminishes the statistical power of such studies and reduces the likelihood of them detecting significant differences between fire treatments. Furthermore, the challenges of achieving prescribed burns that simulate the scale and intensity of wildfires constrain the sample sizes in many experimental studies (e.g. Schlesinger *et al.* 1997; Woinarski 1999; Andersen *et al.* 2005), further diminishing their statistical power (sample sizes of as few as three replicates per treatment). This increases the likelihood of making Type 2 errors, (i.e. concluding that the particular component of the fire regime or some management action under examination has no impact upon the animal or plant in question, when in fact it does). Unfortunately, undiscerning interpretation of such studies can be used as the basis for concluding that the organisms in question are robust or resilient to a wide range of burning regimes. Future experimental studies need to have sufficient genuine replication to ensure that if no effect of the management action is detected, a conclusion that there is no effect is statistically credible.

Sixth, there is an urgent need for landscape-scale space-for-time substitution studies (e.g. Radford and Bennett 2007), where a comparison is made across a large number of study areas, that represent a range of time-since-fire and fire frequency classes and mosaics. Gill (1999) points out that such studies assume 'pre-fire species compositions, fire characteristics and seasonalities of fires were equitable across sites'. While this is true, similar assumptions and limited consideration of the 'invisible mosaic' (Bradstock *et al.* 2005) are also inevitably made when choosing sites for even small-scale experimental studies. While lacking the 'control' of the experimental approach, space-for-time substitution designs can nevertheless provide insights into long-term processes intractable by other methods, and in a much shorter time period. A landscape approach (e.g. Radford and Bennett 2007) also explicitly allows the consideration of landscape-scale issues of patch size, connectivity and the grain size of mosaics.

Seventh, ecologists need to identify more explicit ecological criteria against which the outcomes of fire management can be assessed in each vegetation type. To date, objectives in many management plans are riddled with vague aspirational goals like maintaining or enhancing 'ecosystem health', 'condition' or 'vigour'. Such goals provide no basis upon which to examine the consequences of management actions for fauna, nor refine current practices if they are having adverse effects. Hopefully, future monitoring of the consequences of our actions will be sensitive enough to detect and suggest remedies to address adverse effects before permanent losses occur.

With the gravity of global warming becoming clearer, governments and the public are growing in their appreciation of the pivotal importance of long-term data sets, as we attempt to understand where we have been and where we are headed. Collecting and analysing on-going long-term data on flora *and fauna* from carefully chosen sites and landscapes should become as routine for fire management agencies as collecting meteorological or salinity data is to the agricultural industry. To attempt to operate without it should be as unthinkable as a farmer planting a crop without reference to the rain-gauge or sending sheep to market without first knowing how many will be left in the paddock.

550

Acknowledgements

My deliberations on the issue of managing fire for wildlife have benefited from fruitful discussions with David Baker-Gabb, Andrew Bennett, Kate Callister, David Cheal, Rohan Clarke, Geoff Cary, Ray Damon, Matt Edmunds, Gordon Friend, Peter Green, Brian Malone, David Keith, Greg Jenkins, John Morgan, Neil Murray, Ros St Clair, Kevin Tolhurst, and Neville Walsh, as well as all members of the Mallee Fire and Biodiversity Project Team. I am particularly grateful to Andrew Bennett, David Cheal, Camilla Myers and three anonymous referees for their insightful criticisms of this ms. Nevertheless, the views presented above are personal conclusions, and should in no way be deduced to necessarily represent the views of those named.

560

References

- Abbott, I. (2003) Aboriginal fire regimes in south-west Western Australia: evidence from historical documents. In: 'Fire in ecosystems of south-west Western Australia: impacts and management (Eds I. Abbott and N. Burrows), pp.119-146. (Backhuys Publishers: Leiden.)
- Andersen, A.N., Cook, G.D., Corbett, L.K., Douglas, M.M., Eager, R.W., Russell-Smith, J., Setterfield, S.A., Williams, R.J., and Woinarski, J.C.Z. (2005) Fire frequency and biodiversity conservation in Australian tropical savanna: implications from the Kapalga fire experiment. *Austral Ecology*, **30**, 155-167.
- Baker, J.R. and Whelan, R.J. (1994) Ground Parrots and fire in Barren Grounds, New South Wales: a long-term study and assessment of management implications. *Emu* **94**, 300-304
- Bamford, M.J. (1992) The impact of fire and increasing time after fire upon *Heleioporus eyrei*, *Limnodynastes dorsalis* and *Myobatrachus gouldii* (Anura: Leptodactylidae) in *Banksia* woodland near Perth, Western Australia. *Wildlife Research*, **19**, 169-178.
- Benshemesh, J. (1990). Management of the malleefowl with regard to fire. In 'The mallee lands: a conservation perspective' (Eds J.C. Noble, P.J. Joss and G.K. Jones), pp. 206-211. (CSIRO: Melbourne)
- Benson, J.S. and Redpath, P.A. (1997) The nature of pre-European native vegetation in Australia: A critique of Ryan, D.G., Ryan, J.R. and Starr B.J. (1995). Australian Landscape: observations of explorers and early settlers. *Cunninghamia* **5**, 285-328
- 580 Bradstock R.A., Bedward M., Gill A.M. and Cohn J.S. (2005) Which mosaic? A landscape ecological approach for evaluating interactions between fire regimes, habitat and animals. *Wildlife Research* **32**, 409-423.
- Bradstock, R.A. and Cohn, J.S. (2002) Fire regimes and biodiversity in semi-arid mallee ecosystems. In 'Flammable Australia: the fire regimes and biodiversity of a continent' (Eds. R.A. Bradstock, J.E. Williams and M.A. Gill), pp. 238-258. (Cambridge University Press, Cambridge.)

- Brooker, M.G. (1998) Fire and birds in a West Australian heathland *Emu* **98**, 276-287
- Burbidge, A.H. (2003) Birds and fire in the Mediterranean climate of south-west Western Australia, In 'Fire in ecosystems of south-west Western Australia: impacts and management (Eds I. Abbott and N. Burrows), pp.321-348. (Backhuys Publishers: Leiden.)
- 590 Burgman, M.A. and Lindenmayer, D.B. (1998) 'Conservation Biology for the Australian Environment' (Surrey Beatty and Sons: Sydney.)
- Busby, J.R. (1986) A bioclimatic analysis of *Nothofagus cunninghamii* (Hook.) Oerst. in southeastern Australia. *Austral Ecology*, **11**, 1-7
- Cary, G.J., Flannigan, M., Keane, R., Bradstock, R., Davies, I., Lenihan, J., Li, C., Logan, K., and Parsons, R. (in review) Relative importance of fuel management, ignition management, and weather for area burned: Evidence from five landscape-fire-succession models. *International Journal of Wildland Fire*.
- Caughley, J. (1985). Effect of fire on the reptile fauna of mallee. In 'Biology of Australasian frogs and reptiles' (Eds G. Grigg, R. Shine and H. Ehmann), pp. 31-34. (Surrey Beatty and Sons: Sydney.)
- 600 Chandler, L.G. (1973) In the wake of a bushfire. *Wildlife in Australia*, **10**, 140-141
- Cheal, D. (2004) A consideration of Victoria's proposals for ecological burning, using mallee heathlands as a case study. In 'Proceedings of Bushfire 2004: Earth, Wind and Fire, Fusing the Elements Conference' pp. 1-9. (Department of Environment and Heritage: Adelaide.)
- Christensen, P.E. and Kimber, P.C. (1975) Effects of prescribed burning on the flora and fauna of south-west Australian forests. In 'Managing Terrestrial Ecosystems' (Eds J. Kikkawa and Nix H.A.), pp 85-106. (The Ecological Society of Australia: Brisbane.)
- Cockburn, A., Braithwaite, R.W. and Lee, A.K. (1981) The response of Heath Rat, *Pseudomys shortridgei*, to pyric succession: a temporally dynamic life-history strategy. *Journal of Animal Ecology* **50**, 649-666
- 610 Courtenay, J. and Friend, T. (2004) Gilbert's Potoroo (*Potorous gilbertii*) Recovery Plan July 2003-June 2008. Department of Conservation and Land Management Western Australia, Wanneroo.
- Department of the Environment and Water Resources (2007) 'Australia's Native Vegetation: A Summary of Australia's Major Vegetation Groups, 2007'. (Australian Government: Canberra)
- Department of Sustainability and Environment (2006). Code of Practice for Fire Management on Public Land. Victorian Government Department of Sustainability and Environment, Melbourne.
- Department of Sustainability and Environment (2007) Mildura District Fire Protection Plan. Victorian Government Department of Sustainability and Environment, Melbourne.
- Dostine, P.L., Johnson, G.C., Franklin, D.C. and Hempel, C. (2001) Seasonal use of savanna landscapes by the Gouldian finch *Erythrura gouldiae*, and co-existing finches in the Yinberrie Hills area, Northern Territory. *Wildlife Research* **28**, 445-458
- 620 Edwards, A., Kennett, R., Price, O., Russell-Smith, J. Spiers, G. and Woinarski, J. (2003) Monitoring the impacts of fire regimes on vegetation in northern Australia: an example from Kakadu National Park. *International Journal of Wildland Fire* **12**, 427-440
- Fernandes, P.M. and Botelho, H.S. (2003) A review of prescribed burning effectiveness in fire hazard reduction. *International Journal of Wildland Fire* **12**, 117-128
- Fire Ecology Working Group (2002). Analysis of disturbance by fire on public land in Victoria. (Department of Natural Resources and Environment: Melbourne.)

- Fire Ecology Working Group (2004). Guidelines and Procedures for Ecological Burning on Public Land in Victoria 2004. (Department of Sustainability and Environment, Melbourne.)
- 630 Fox, B.J. (1982) Fire and mammalian secondary succession in an Australian coastal heath. *Ecology* **63**, 1332-1341
- Fox, B.J. (1990) Changes in the structure of mammal communities over successional time scales *Oikos*, **59**, 321-329
- 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. (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'. Pp. 155-161. (Department of the Environment, Sport and Tourism: Canberra)
- 640 Gibbons, P. and Lindenmayer, D.B. (1997). Conserving hollow-dependent fauna in timber-production forests. (NSW National Parks and Wildlife Service: Hurstville.)
- Gill, A.M. (1975) Fire and the Australian flora. *Australian Forestry*, 4-25.
- 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). (Australian Academy of Science: Canberra.)
- Gill, A.M. (1999). Biodiversity and bushfires: an Australia-wide perspective on plant-species changes after a fire event. In 'Australia's Biodiversity- responses to fire: plants, birds and invertebrates' (Eds A.M. Gill, J.C.Z. Woinarski and A. York), pp. 9-54. (Department of the Environment and Heritage: Canberra).
- 650 Gill, A.M., Bradstock, R.A., and Williams, J.E. (2002). Fire regimes and biodiversity: legacy and vision. In 'Flammable Australia: the fire regimes and biodiversity of a continent' (Eds. R.A. Bradstock, J.E. Williams and A.M. Gill), pp. 429-446. (University Press, Cambridge.)
- Gill, A.M. and McCarthy, M.A. (1998) Intervals between prescribed fires in Australia: what intrinsic variation should apply? *Biological Conservation*, **85**, 161-169.
- Good, R. (1981). Adaptations of Australian plants to fires. In 'Bushfires: their effect on Australian Life and Landscape' (Ed. P. Stanbury), pp. 48-59. (The Macleay Museum: Sydney.)
- Good, R.B. (1996). Fuel dynamics, preplan and future research needs. In 'Fire and Biodiversity - The Effects and Effectiveness of Fire Management'. Pp. 253-266. (Department of the Environment, Sport and Tourism: Canberra).
- 660 Griffiths, A.D. and Christian, K.A. (1996) The effects of fire on the frillneck lizard (*Chlamydosaurus kingii*) in northern Australia. *Australian Journal of Ecology*, **21**, 386-398.
- Johnson, E.A. and Gutsell, S.L. (1994) Fire frequency models, methods and interpretations. *Advances in Ecological Research*, **25**, 239-287.
- Jurskis, V. (2005) Decline of eucalypt forests as a consequence of unnatural fire regimes. *Australian Forestry* **68**, 257-262
- Keith, D.A. and Henderson, M.K. (2002) Restoring the prepastoral condition: response to Jurskis. *Austral Ecology*, **27**, 691-693.
- Keith, D.A., Williams, J.E., and Woinarski, J.C.Z. (2002). Fire management and biodiversity conservation: key approaches and principles. In 'Flammable Australia: the fire regimes and biodiversity of a continent' (Eds. R.A. Bradstock, J.E. Williams and M.A. Gill), pp. 401-425. (Cambridge University Press, Cambridge.)
- 670

- Kenny, B., Sutherland, E.F., Tasker, E., and Bradstock, R. (2004). Guidelines for ecologically sustainable fire management. NSW biodiversity strategy. (Biodiversity Research and Management Division (Bushfire Research), NSW National Parks and Wildlife Service: Hurstville.)
- Koch, P. (2005) 'Factors influencing food availability for the endangered south-eastern Red-tailed Black Cockatoo *Calyptorhynchus banksii graptogyne* in remnant stringbark woodland, and implications for management', PhD Thesis, University of Adelaide.
- Lambeck, R.J. (1997) Focal species: a multi-species umbrella for nature conservation. *Conservation Biology*, **11**, 849-856.
- 680 Letnic, M. and Dickman, C.R. (2005) The responses of small mammals to patches regenerating after fire and rainfall in the Simpson Desert, central Australia. *Austral Ecology* **30**, 24-39
- Letnic, M., Tamayo, B. and Dickman, C.R. (2005) The responses of small mammals to La Nina (El Nino Southern Oscillation)-associated rainfall, predation, and wildfire in arid Australia. *Journal of Mammalogy* **86**, 689-703
- Mackey, B., Lindenmayer, D.B., Gill, M., McCarthy, M., and Lindsay, J. (2002) 'Wildlife, Fire and Future Climate: A Forest Ecosystem Analysis' (CSIRO: Collingwood.)
- Mackowski, C.M.(1984) The ontogeny of hollows in blackbutt, *Eucalyptus pilularis* and its relevance to the management of forests for possums, gliders and timber. In 'Possums and Gliders' (Eds. A. P. Smith and I.D. Hume) pp 517-525. (Surrey Beatty and Sons: Sydney.)
- 690 Masters, P. (1993) The effects of fire-driven succession and rainfall on small mammals in spinifex grassland at Uluru National Park, Northern Territory. *Wildlife Research* **20**, 803-813
- McCarthy, G.J., Tolhurst, K.G., and Chatto, K. (1999). Overall Fuel Hazard Guide. (Department of Natural Resources and Environment: Melbourne.)
- Meredith, C.W., Gilmore, A.M. and Isles, A.C. (1984) The ground parrot (*Pezoporus wallicus* Kerr) in south-eastern Australia: a fire-adapted species? *Australian Journal of Ecology* **9**, 367-389
- Morrison, D.A. Buckney, R.T., Bewick, B.J. and Cary, G.J. (1996) Conservation conflicts over burning bush in south-eastern Australia. *Biological Conservation* **76**, 167-175
- Nield, C. (2006) Fire mosaics: The influence of variations in fire severity on initial post-fire biotic responses after the 2005 wildfire at Wilsons Promontory. Hons Thesis, Department of Botany, La Trobe University.
- 700 Noble I.R. (1982) Ecological effects of fire – an overview. In 'Fire Ecology in Semi-arid Lands' (Eds. A. Heislars, P. Lynch and B. Walters) pp (CSIRO Division of Land Resources Management Communications Group: Deniliquin.)
- Noble, I.R. and Slatyer (1980) The use of vital attributes to predict successional changes in plant communities subject to recurrent disturbances. *Vegetatio*, **43**, 5-21.
- Noble, J.C. and Vines, R.G. (1993) Fire studies in mallee (*Eucalyptus* spp.) communities of western New South Wales: grass fuel dynamics and associated weather patterns. *Rangeland Journal*, **15**, 270-297.
- 710 Noss, R.F. (1990) Indicators for measuring biodiversity: A hierarchical approach. *Conservation Biology*, **4**, 355-364.
- O'Brien, T.P. (1989). The impact of severe frost. In 'Mediterranean Landscapes in Australia Mallee Ecosystems and their Management'(Eds J.C. Noble and R.A. Bradstock) (CSIRO: 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
- Quinn, G.P. and Keogh M.J. 'Experimental Design and Data Analysis for Biologists' (Cambridge University Press: Cambridge)
- Radford, J.Q. and Bennett, A.F. (2007) The relative importance of landscape properties for woodland birds in agricultural environments. *Journal of Applied Ecology*, **44**, 737-747
- 720 Schlesinger, C.A., Noble, I.R., 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. *Rangelands Journal*, **19**, 190-205.
- Tolhurst, K.G. and Friend, G.R. (2001) An objective basis for ecological fire management. In 'Bushfire 2001 Proceedings of the Joint Bushfire/FRFANZ Conference'. pp. 220-228. (New Zealand Forest Research Institute Ltd: Rotorua)
- Ward, M.J. and Paton, D.C. (2004) Responses to fire of Slender-billed Thornbills, *Acanthiza iredalei hedleyi* in Ngarkat Conservation Park, South Australia. I. Densities, group sizes, distribution and management issues. *Emu*, **104**, 157-167.
- Whelan, R.J. (1995) 'The Ecology of Fire' (Cambridge University Press: Cambridge).
- 730 Whelan, R.J., Rodgerson, L., Dickman, C.R., and Sutherland, E.F. (2002). Critical life cycles of plants and animals: developing a process-based understanding of population changes in fire-prone landscapes. In 'Flammable Australia: the fire regimes and biodiversity of a continent' (Eds R.A. Bradstock, J.E. Williams and A.M. Gill), pp. 94-124. (Cambridge University Press: Cambridge.)
- Williams, R.J., 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.
- Williams, R.J., Woinarski, J.C.Z., and Andersen, A.N. (2003) Fire experiments in northern Australia. contributions to ecological understanding and biodiversity conservation in tropical savannas. *International Journal of Wildland Fire*, **12**, 391-402.
- 740 Willson, A. (1999) Tarawi Nature Reserve Fire Management Plan. (NSW National Parks and Wildlife Service: Hurstville.)
- Woinarski, J.C.Z. (1999). Fire and Australian birds: a review. In 'Australia's biodiversity: responses to fire' (Eds A.M. Gill, J.C.Z. Woinarski and A. York) pp. 55-111. (Department of Environment and Heritage: Canberra.)
- Woinarski, J.C.Z. (2005). Living with fire - birds in northern Australia. In Fire and birds: fire management for biodiversity. (Eds P. Olsen and M. Weston) pp 7-9 Supplement to Wingspan 15) (Birds Australia: Hawthorn East.)
- 750 Woinarski, J.C.Z., Armstrong, M., Price, O., McCartney, J., Griffiths, A.D. and Fisher, A. (2004) The terrestrial fauna of Litchfield National Park, Northern Territory: monitoring over a 6-year period and response to fire history. *Wildlife Research*, **31**, 587-596
- Woinarski, J.C.Z., Brock, C., Fisher, A., Milne, D. and Oliver, B. (1999) Response of birds and reptiles to fire regimes on pastoral land in the Victoria River District, Northern Territory. *Rangeland Journal*, **21**, 24-38
- Woinarski, J.C.Z., Williams, R.J., Price, O., and Rankmore, B. (2005) Landscapes without boundaries: wildlife and their environments in northern Australia. *Wildlife Research*, **32**, 377-388.

Wouters, M.A. (1993). Developing Fire Management Planning in Victoria: A case study from the Grampians. (Department Conservation and Natural Resources: Melbourne.)

760 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' (Eds A.M. Gill, J.C.Z. Woinarski and A. York) pp. 181-266. (Department of Environment and Heritage: Canberra.)