**ABSTRACT**

We discuss interactions between fire regimes and biodiversity in the *Eucalyptus* forests and woodlands of temperate south-eastern Australia, and the savannas of wet-dry tropical northern Australia. Despite the major geomorphic and climatic differences between these regions, we argue that in both parts of the country, fire regimes and the fire ecology of species are governed by a common set of factors: (1) the incidence of severe fire weather, (2) ignition probability, (3) fuel accumulation, and (4) critical life history processes (e.g., juvenile periods, acquisition of fire tolerance, dispersal). The relationships between fire regimes and biodiversity (i.e., the persistence of species at landscape scales) will reflect differences in these processes between regions.

In both regions, however, a major management question concerns fires with a return time of 1–10 years, in terms of their effects on both fuel and the biota. In the savannas, fires are frequent and spatially extensive, but of relatively low intensity. Fuel reduction burning is prescribed as a general management tool, primarily to minimize the extent and impact of late dry-season fires, which can have deleterious impacts on plants and animals in the savanna. The use of prescribed fire thus has an explicit biodiversity component in the management of landscapes in northern Australia. In temperate Australia, fires are relatively infrequent, and potentially very intense. Fuel reduction burning is used and promoted as a general management tool, primarily for the protection of life and property, rather than the maintenance of biodiversity. Extensive use of prescribed fire is necessary to achieve maximum levels of human protection in south-eastern Australia, but such a regime could be incompatible with the conservation of biodiversity. Although there are clearly species and communities in both regions that can persist in the landscape in the face of frequent fires, some plant functional groups, and some small mammals, are at risk in the face of inappropriate fire regimes. Further research is required to integrate point-based studies of fire impacts with a spatial understanding of ignitions, fire behavior, and ecosystem dynamics.

*keywords*: Australia, biodiversity, *Eucalyptus*, fire regimes, forest, fuel accumulation, national park, savanna, temperate, tropical, woodland.

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**INTRODUCTION**

The landscapes of the Australian continent as a whole are both fire-prone and biologically diverse (Gill et al. 1981, Groves 1994). A fundamental scientific and management question, therefore, is identifying and incorporating the most appropriate fire regimes into the management of this precious biological diversity. We highlight below some of the problems in arriving at an agreed-upon set of appropriate fire regimes at the landscape scale for 2 distinctive and extensive Australian forest types: (1) eucalypt woodlands and open forests with a shrubby understory from the temperate south-east, and (2) eucalypt savanna woodlands with a grassy understory from the wet-dry tropics of the north.

The regions we discuss, though several thousand kilometers apart, have interesting parallels, and offer the potential for developing common principles in fire and biodiversity management. Both contain diverse eucalypt-dominated forests and woodlands that are represented in extensive and high profile National Parks close to urban centers (e.g., Morton National Park near Sydney, and Kakadu National Park near Darwin). A major fire issue in both regions is the effect of frequent fire (at 1–10-year return intervals) on plant and animal biodiversity, and on fuel reduction and thus the incidence of unwanted fires. We have restricted our discussion to the more open eucalypt-dominated communities, which are common to both regions. This does not imply, however, that other forest vegetation types such as rainforest and tall eucalypt forests (sensu Groves 1994) do not present problems in fire management, or that fire frequency is not an issue for biodiversity conservation.

**A FRAMEWORK FOR ASSESSING FIRE REGIMES AND BIODIVERSITY INTERACTIONS IN AUSTRALIA**

Despite major differences in climate and soils across Australia, we argue that fire regimes and their
impacts on ecosystems in different regions are governed by a common set of factors. These differ from place to place (e.g., tropical versus temperate zones, mesic versus arid environments) in rate or extent, such that each place has a unique combination of factors. The critical processes are: (1) annual incidence of severe fire weather; (2) rates of ignition, both seasonal and annual; (3) rates of fuel accumulation; and (4) critical life history processes, e.g., time to flowering, length of juvenile periods, acquisition of fire tolerance, and dispersal (Whelan 1995, Bond and van Wilgen 1996, Keith 1996).

These factors affect individual fires (e.g., size and shape), resultant fire regimes (e.g., intensity and frequency of fires), and the likelihood of persistence of species at landscape scales. Different outcomes may be expected across landscapes of differing topographic complexity (e.g., size and connectivity of fragments) even within a single region. The nature of land use and administrative boundaries further influences outcomes.

Our thesis is that these fire, biotic, and landscape variables can be used systematically to examine fire regimes and their biotic consequences. By contrasting the nature of fire regimes and biodiversity between tropical and temperate regions, we intend to shed light on how management through prescribed burning can be focused to achieve multiple land use objectives.

Fire in Tropical and Temperate Landscapes

The Geographical Setting

The savannas of the Darwin region (ca. 12°S; 131°E) are the dominant vegetation type in the higher rainfall regions of northern Australia, where annual rainfall exceeds 1,000 millimeters. They typically occur on light-medium textured soils (sands and loams) on generally flat land surfaces. They are characterized by a tree stratum 15–20 meters tall, and an understory that is usually dominated by species of the tall, annual grass *Sorghum* (Wilson et al. 1990). The savannas are the “sea” in which non-woodland vegetation occurs, such as treeless grasslands and some heath communities.

Forests and associated woodlands in the Sydney region of south-eastern Australia (ca. 34°S; 151°E) vary greatly in terms of structure and floristic composition (Gill 1994). The spectrum of forest types corresponds to moisture and fertility gradients. Site conditions, as determined by terrain and parent material, determine the overall mix of forest types. Much of the vegetation is strongly influenced by the predominant sandstone geology. The resultant landscape is dissected, with abrupt valleys and canyons and expanses of mainly sandy soils of varying depth but generally low fertility. Heaths, shrublands and woodlands, and low-open forests with a heathy understory are found, often in complex arrays according to soil depth, drainage, and aspect. A common feature of these communities is a prominent shrub layer containing a diverse array of species, with different life forms and regeneration strategies (Benson and Howell 1990, Benson and McDougall 1996). Rainforest or tall eucalypt forest may be found in deep moist gullies but also in other topographic situations in association with igneous intrusions or beds of shale. The structure, composition, and positioning of these vegetation types is likely to have been similar over the past 10,000 years (Benson and Redpath 1997).

Fig. 1. Seasonal 1500-hour Forest Fire Danger Index (FFDI) for Jabiru, 250 kilometers east of Darwin, Northern Territory. MAXFFDI = absolute maximum FFDI for month; MMAXFFDI = mean maximum FFDI for month. From Gill et al. (1996).

Fire Weather, Ignition, and Historical Changes to Ignition

Several fire danger rating systems are used in Australia, based on standard fire weather variables and fuel moisture. The most common is McArthur’s Forest Fire Danger Index (FFDI) (Luke and McArthur 1978, Noble et al. 1980). FFDI ranges from 0–100, with values >50 indicating extreme fire weather.

Fire weather in tropical savannas is governed primarily by the arrival and departure of the monsoon (Gill et al. 1996). During the peak monsoon period (Jan—early Mar), when the majority of the year’s rain falls, the FFDI is <10 (Figure 1) and the vegetation is essentially non-flammable. From late March onward, both atmospheric and soil moisture decline rapidly (Gill et al. 1996). Nevertheless, average maximum FFDI in the early dry-season (May–Jun) remains <30, steadily increasing to values of around 40 in September. The most extreme value recorded in 11 years’ records at Jabiru (250 kilometers east of Darwin) was ca. 60, well below peak levels of ca. 90, which occur on extreme days in the Sydney region (Gill and Moore 1994, 1996).

Fire is frequent in Australia’s savannas. In the Adelaide Rivers—Kakadu region, Press (1988), Braithwaite and Estbergs (1985), and Russell-Smith et al. (1997) reported fire frequencies of approximately 1 year in two or 2 years in three. This high fire frequency results from the combination of predictable rainfall, consistent production of biomass in the understory every wet season, a 5–8-month dry season that cures fine fuels every year, the generally inter-annual predictability in fire weather, and, most importantly, high rates of ignition, which, at present, is from human sources (Gill et al. 1996, Williams et al. 1998). Prescribed fire is a landscape management tool for vir-
Fire regimes and the management of biodiversity in *Eucalyptus*

Fire weather in the Sydney region, like most of south-eastern Australia, is influenced by the passage of frontal systems traveling generally in an easterly direction (Speer et al. 1996). These fronts result in periods of strong winds (>50 kilometers per hour), which can shift rapidly from north to west and south. Prior to the passage of fronts, high temperatures and low relative humidity (<20%) may occur as northerly winds draw warm, dry air across from the interior of the continent. Analysis of fire weather over a 40-year period (Figure 2) revealed that for the vast majority of the period FFDI was <10. There was on average 1 Extreme day (FFDI > 50) per year (Gill and Moore 1994, 1996).

Most fires are attributable to human sources, with lightning causing <1% of fires in this region (New South Wales [NSW] National Parks and Wildlife Service, unpublished records). Significant lightning-caused fires associated with severe drought currently occur on about a 20–30-year cycle (NSW National Parks and Wildlife Service, unpublished records). Importantly, the incidence of fire appears to be related to FFDI. In major conservation reserves in the Sydney region, the proportion of days on which unplanned fires occurred was positively related to 1500-hour FFDI (Figure 3). A similar trend was found by Gill et al. (1987) across the state of Victoria. Most unplanned fires in these reserves are attributable to arson or negligence (Conroy 1996, NSW National Parks and Wildlife Service, unpublished records). However, it is not known whether this trend is due to more ignitions under more extreme fire weather conditions, or a greater chance of successful ignition and spread.

Fire regimes have probably shifted in both regions since prehistoric times. In the savannas, before Aboriginal people arrived, fire probably occurred during the transition period between the dry and wet seasons. This period is the only one during the year that has both dry, combustible fuel, and lightning. Aboriginal people modified the timing of fire by burning during the dry season. Fires commenced in the early dry-sea-son (March), increased in frequency until about July, decreased in frequency until the end of the dry season, then increased again (Braithwaite 1991). The incidence of late dry-season fires appears to have increased following European settlement of the Top End of the NT until the end of the 1980's (Press 1988). In Kakadu National Park, however, late dry-season fires appear to have occurred less frequently since the early 1980's (Russell-Smith et al. 1997).

In south-eastern Australia, quantitative measures of changes in the incidence of fire are not available generally, although a number of case studies in the mountainous country are available (Williams and Gill 1995) mainly through dendrochronological methods. These studies indicate distinct and differing periods of fire activity after European settlement began. Evidence suggests that in some cases the incidence of fire increased in association with grazing activities (e.g., Leigh et al. 1987). In the mid-twentieth century the incidence of fire subsequently declined. The general applicability of this scenario is unknown, but unpublished records in the Sydney region indicate highly variable trends in the overall incidence of fire (both planned and unplanned) in the last 20 years.

**Fuels**

Fuel in the tropics is produced consistently each year. Most of the fine fuels are grass. The proportion of leaf and twig components from both evergreen and deciduous trees increases as the dry season progresses (Williams et al. 1997). Annual fine fuel production is of the order of 3–5 metric tons per hectare, and fuel loads tend to remain within this domain with annual burning. Equilibrium fuel loads are of the order of 8–10 metric tons per hectare, which can be achieved within 2–3 years without fire (Figure 4) (Cook et al. 1995). Thus, fine fuels are produced each year to levels that can sustain fire, and can effectively reach near-maximal levels in 2 or 3 years without fire.

In contrast, equilibrium fuel loads in temperate vegetation communities around Sydney are of the order of 20–30 metric tons per hectare (Figure 5), reached 10–20 years after burning. Heaths and shrublands show the highest rates of accumulation with a substantial component of fine fuel being contributed...
by live and dead shrub crowns. Litter fuels are higher in woodlands and low-open forest than in heath. Morrison et al. (1995) found that substantial quantities of fine fuel (3–4 metric tons per hectare) were present immediately after low-intensity prescribed fires, as a consequence of both incomplete fuel consumption and scorched leaves falling to the ground. In contrast, there was little or no fuel left after high-intensity fires. Thus, in the sandstone communities, fuel accumulation following low-intensity fire is sufficient for fires to propagate under severe weather 2–3 years after burning (e.g., Bradstock and Scott 1995, Conroy 1996). The time for fuel to accumulate to levels sufficient for uncontrollable fire behavior under severe weather is also relatively short, i.e., 2–4 years (Morrison et al. 1995, Bradstock et al. 1998b).

Fire Behavior and Size of Fires

Fires in tropical savannas are of relatively low intensity. Byram fire intensity is generally 500–10,000 kilowatts per meter. Fires >20,000 kilowatts per meter, with crown fires and spot fires up to several hectares ahead of the main front, do not occur (Williams et al. 1998). There is a distinct seasonality to the fire regime. Fire weather is most extreme in the latter part of the dry season (Sep–Oct). Fuel loads increase from the early dry to the late dry-season, due to tree leaf litter inputs (Williams et al. 1997). Hence, late dry-season fires are, on average, 3–4 times more intense than those of the early dry-season fires—about 2,000 kilowatts per meter compared with 8,000 kilowatts per meter (Williams et al. 1998). Fires also tend to be smaller in early dry season than in late dry season—about 50–100 hectares compared with several thousand hectares (Russell-Smith et al. 1997).

In contrast to the savannas, fires in southern temperate low open forests and woodlands may be >20,000 kilowatts per meter, reflecting the potentially higher fuel loads, and the higher wind speeds associated with severe weather. Such fires may lead to complete consumption of crowns in trees and long-distance spotting in severe weather (Luke and McArthur 1978, Cheney 1981, Gill 1997). Under conditions when FFFI > 50 (winds speed typically >30 kilometers per hour), headfires are observed to spread at rates of 2–4 kilometers per hour (Conroy 1996, National Parks and Wildlife Service, unpublished data). Fires 1,000–10,000 hectares in area may occur under such conditions and have the potential to burn a large portion of medium-sized conservation reserves in 1–5 days. Often the problem of such relatively large fires is exacerbated by back-burning operations intended to provide containment in inaccessible terrain. Some evidence (e.g., Conroy 1987) suggests that the size of fires is exponentially related to factors such as the drought index. A summary of the broad differences in the fire weather, fuels, and fire characteristics of the Sydney and Darwin regions is given in Table 1.

Fire Ecology and Critical Life-history Processes

The savannas exhibit a high degree of resilience to fire at the landscape level. Most of the constituent perennial plants, both woody and herbaceous, can resprout vegetatively following fire. All of the eucalypts are resprouters, rather than seeders (Lacey 1974), which is an adaptation to seasonal drought as much as to fire (Gill 1997). Reestablishment of the canopy in the dominant eucalypts is usually complete by the end

Table 1. A comparison of the broad fire weather, fuel, and fire behavior characteristics of the landscapes of the Darwin Region and the Sydney Region.

<table>
<thead>
<tr>
<th></th>
<th>Darwin</th>
<th>Sydney</th>
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<tbody>
<tr>
<td>Peak fire weather period</td>
<td>September/October</td>
<td>November</td>
</tr>
<tr>
<td>Maximum FFFI</td>
<td>60</td>
<td>100</td>
</tr>
<tr>
<td>Equilibrium fuel load</td>
<td>10 metric tons per hectare</td>
<td>30 metric tons per hectare</td>
</tr>
<tr>
<td>Major fuels</td>
<td>Grass</td>
<td>Leaf and twig</td>
</tr>
<tr>
<td>Ladder fuels</td>
<td>No</td>
<td>Yes</td>
</tr>
<tr>
<td>Maximum fire intensity</td>
<td>20,000 kilowatts per meter</td>
<td>50,000 kilowatts per meter</td>
</tr>
<tr>
<td>Landscape relief</td>
<td>10–100 meters</td>
<td>50–500 meters</td>
</tr>
<tr>
<td>Crown fires</td>
<td>No</td>
<td>Yes</td>
</tr>
</tbody>
</table>
of the wet season following fire (R.J. Williams, unpublished data). Adult eucalypts and other canopy subdominants can flower in the season following fire (Williams 1997). The eucalypts are non-serotinous; seed release occurs each year, as fruits ripen and dry, rather than periodically in response to fire, as is common in south-eastern Australia (Dunlop and Webb 1991, Setterfield and Williams 1996). Importantly, there are very few species of obligate seeders or trees in tropical savannas, e.g., <1% of the woody species of the savannas of Kakadu National Park (Brennan 1996). The 1–2 year fire-return period, coupled with the rapid postfire responses of the vegetation, means that gradual, postfire successional patterns in the vertebrate fauna, which are typical of the fire responses of many vertebrates in southern Australia, do not occur in the savannas (Braithwaite and Werner 1987, Woinarski 1990).

Open forests and woodlands of south-eastern Australia contain a wide array of plant and faunal functional groups with a correspondingly wide spectrum of fire regime requirements. Shifts in fire regimes at both the point and landscape scales will affect diversity and structure. Typically, overstory Eucalyptus and Angophora species of either single-stemmed or multi-stemmed habit (mallees) can resprout from a basal lignotuber. Arborescent species of Eucalyptus may lose this basal resprouting capability once trunks develop, relying instead on insulated epicormic stem buds to initiate crown recovery after loss through fire. Similar modes of recovery occur in many of the understory shrubs. Although eucalypts that lack the capability to resprout are generally absent from low-open forests and woodlands in south-eastern Australia, many such species of understory shrubs, herbs, and sedges are common in these communities and may constitute up to 50% of the flora (e.g., Nieuwenhuis 1987, Morrison et al. 1995, 1996, Bradstock et al. 1997).

Reproductive and seedbank characteristics vary among strata in the vegetation. Tree species typically retain seeds in woody capsules for ≥1–2 years (Lamont et al. 1991, Pook et al. 1997). Adult trees of Eucalyptus and Angophora species are usually able to flower within 2–3 years of fire even after sustaining extensive crown damage (Gill 1997). Similar flowering patterns have been observed in resprouting shrubs. Obligate seeder shrub and herb species exhibit a similar range of seedbank characteristics. Species with on-plant storage of seeds in serotinous fruits, and within-soil storage are particularly common (Keith 1996). Fire triggers germination either through the release of the aboveground seedbank or through cues that break the dormancy of buried seed. Species incapable of resprouting depend on such mechanisms for recovery following fire.

The juvenile period of species, whether resprouting or obligate seeders, is important. Typically, establishment of seedlings is linked to fire as a result of direct and indirect stimuli and other effects such as predator satiation (O’Dowd and Gill 1984, Bond and van Wilgen 1996). However, recurrence of fire during the juvenile period may result in the immediate decline or elimination of a population. The development of a fire-resistant lignotuber, capable of vegetative regrowth after fire, determines the minimum interval between fires that can be sustained by populations in the long-term. If fire recurs before the lignotubers of new recruits are sufficiently well developed, they will die. Typical juvenile periods of obligate seeders in the flora of the Sydney region are 2–4 years for legumes, and 4–10 years for many species of Proteaceae, Allocasuarina, and Callitris (Benson 1985, Keith 1996).

The prominence of shrub species in woodlands and forests of south-eastern Australia is of twofold significance for biodiversity and fire management. First, many species are characteristic of, or endemic to, the area. They have conservation significance in their own right. Second, the shrub layer contributes to habitat for many animals. For mammals the prevailing paradigm in south-eastern forests and woodlands is the positive relationship between habitat complexity, and the abundance and diversity of animals (Catling 1991, Catling and Burt 1995, Cork and Catling 1996). If the understory is diminished or removed, the abundance and diversity of mammals may decline. High frequencies of fire can result in such habitat simplification by disrupting plant regeneration processes.

Changes in diversity, richness, and abundance of invertebrates in relation to changes in forest structure have also been described (York 1995). Complex vegetation contains specialist species of invertebrates not present in habitat with open structure. Thus, structurally complex forests may offer a greater range of habitat conditions for animal species than forests with simpler structure.

THE PRESCRIBED BURNING AND BIODIVERSITY PROBLEM

Tropical Eucalypt Savannas

The savannas are extremely rich in flora and fauna (Braithwaite and Werner 1987, Bowman et al. 1993, Brennan 1996). Within the mesic savannas, diversity across virtually all trophic levels is higher in the savannas than in monsoon rainforest, or grasslands of the fertile cracking clay soils of the floodplains (Williams et al. 1995). The World Heritage Kakadu National Park contains about 1,800 species of vascular plants, >33% of the 4,500 species for the NT as a whole (Brennan 1996). Historically, Australian savannas are unlikely to have been derived by recent (late Pleistocene) burning of rainforest (Bowman 1999) and are likely to have always been both rich biologically, and fire prone. Australia’s savannas are also relatively intact ecologically (Williams et al. 1996) because forestry and intensive agriculture have been of limited extent both spatially and historically. The savannas also represent important habitats for several groups of vertebrates whose range and/or abundance have contracted since European settlement of Australia (Woinarski and Braithwaite 1990).

Given the 2–3 year return time of fires in the landscape, and the increase in potential intensity as the dry
season progresses, management questions regarding fire regimes and biodiversity in the savannas have to do with both fire frequency and fire intensity. The apparent increase in the incidence of extensive, relatively intense late dry-season fires since European occupation is of particular concern, and the prevention of such fires, especially repeat late dry-season fires, is a general management objective (Russell-Smith 1995, Russell-Smith et al. 1997).

The impact of late dry-season fires, both as individual fires and repeat fires, can be dramatic. Their effects are most profoundly displayed in the tree stratum. Individuals of fire-sensitive species, such as native cypress (C. intratropica) die following complete canopy scorch. Callitris woodlands have contracted in area in the past 50–100 years because of an increased incidence of late dry-season fires (Bowman and Panton 1993). Some vegetation types, such as monsoon rainforest (Bowman 1991) and heathlands with a high proportion of obligate seeder species (Russell-Smith et al. 1997) may undergo local contraction or substantial changes in species composition under a regime of repeated, extensive late dry-season fires.

Within the eucalypt-dominated savannas, species vary in susceptibility to late dry-season fires (Lonsdale and Braithwaite 1991, Williams 1995, Williams and Cook 1998, Williams et al. 1999). The deciduous, broad leaf trees (e.g., Terminalia), and the bloodwood group of eucalypts (e.g., E. porrecta) are more sensitive to intense, late dry-season fires than are the dominant eudensid group of eucalypts (e.g., E. miniata). Late dry-season fires also reduce the floral and fruit reserves across all functional groups of trees (Setterfield 1997, Williams 1997). This reduction has implications not only for tree regeneration and diversity, but also for fauna, such as mobile nectarivorous birds, which depend upon the floral resources of trees, especially in the dry season (Franklin 1997). Granivorous birds may also be disadvantaged by late dry-season fires because such fires may reduce the reserves of grass seeds (Woinarski 1990). The nutrient capital of the savanna may also be reduced by late dry-season fires, via the incremental effect on tree mortality, and therefore biomass (Cook 1994, Hurst et al. 1996).

Some elements of the landscape do not appear to suffer unduly from late dry-season fires. The composition of ground stratum vegetation was little different between late-burned, early-burned, and unburned treatments over a 14-year period at Munmarlary (Bowman et al. 1988). A similar pattern for these regimes occurred at Kapalga over a 5-year period (R.J. Williams, unpublished data). Late dry-season fires benefit some faunal groups, such as arid-adapted groups of ants (Andersen 1991). Late dry-season fire may also increase nutrient pulses into small streams at the commencement of the wet season, with consequent increases in the diversity of aquatic invertebrate biota (M. Douglas, NT University, and S. Townsend, NT Power and Water Authority, unpublished data). Even frill-neck lizards, which can suffer 30% mortality during late dry-season fires, are not unduly disadvantaged by late dry-season fires because compensating factors such as extra food resources allow additional reproduction and recruitment (Griffiths and Christian 1996).

How do the impacts of early dry-season fires compare with those of late dry-season fires? Evidence is accumulating, especially from the Munmarlary and Kapalga fire experiments (Bowman et al. 1988, Andersen et al. 1998), that early dry-season fires are either relatively benign or indeed may benefit some elements of the biota. These studies have indicated relatively few impacts on understory composition (Bowman et al. 1988), tree mortality (Williams and Cook 1998, Williams et al. 1999), tree reproductive phenology (Williams 1997), and water quality (Townsend 1997). Some elements of the fauna, such as arid-adapted ants (Andersen 1991), herpetofauna in general (Trainor and Woinarski 1994), and frill-neck lizards in particular (Griffiths and Christian 1996) benefit from early dry-season fires. There is also a broad suite of birds in savannas that show short-term positive responses to fire. These species are attracted to recently burned areas and are not necessarily disadvantaged by frequent fire (Woinarski 1990).

However, some evidence suggests that for certain groups and landscape processes, annual early dry-season prescribed fires may not be so benign. Such fires may reduce seed output in one of the dominant eucalypts (E. miniata) (Setterfield 1997), and limit seedling recruitment in this species and another common eucalypt, E. tetrodonta. The abundance of some guilds of small mammals, including rare and endangered species for which northern Australia represents an extremely important refuge, may be reduced by repeat early dry-season fire. This species would appear to require protection from fire for about 2–3 years for the maintenance of population structure, and interspecific diversity (R.W. Braithwaite, unpublished data). Several species of birds are disadvantaged by high-frequency fire, but they persist in the landscape by exploiting the mosaic nature of the landscape, by dispersing into unburned areas (Woinarski 1990).

Frequent fire, including low-intensity early dry-season fire, may represent a bottleneck to recruitment (Werner 1986). Fensham and Bowman (1992) argued against this process, however, on the basis of stand structure in the tall mesic savannas on deep soils on Melville Island. Where site quality is low, perhaps because of low-nutrient soils, or seasonal inundation, then frequent fire may suppress woody species, and prevent the development of a tree stratum (Wilson and Bowman 1993).

Fire exclusion in savannas may affect biodiversity. There are some, albeit rare, patches of savanna that have been unburned for 5–20 years. Such decadal absence of fire appears to result in a more diverse tree stratum (Bowman et al. 1988, Fensham 1990, Gill et al. 1990, Braithwaite 1995a, b). However, the absence of fire for 10 years does not appear to result in a succession towards closed rainforest, unlike some savanna areas in Africa (Bowman et al. 1988, Menaut et al. 1996). In contrast to tree diversity, herb diversity may decline in the longer-term absence of fire (Fensham 1990).
Excluding fire from year to year in particular patches of the mesic savannas is a potential fuel management problem, given that fuel loads can effectively double from 2–5 metric tons per hectare, to near-maximal loads of 5–10 metric tons per hectare within 2–3 years of fire exclusion (Gill et al. 1990, Cook et al. 1995). If, within a given patch of savanna, the specific management goal includes short term (2–3 year) and longer-term (decadal) protection from fire, then consideration must be given to the consequences of a predictable, rapid accumulation of fuel to near-maximal levels in 2–3 years. This accumulation of fuel can pre-dispose the savanna to late dry-season, high-intensity fires, which are likely to occur eventually in such fire-protected sites. This scenario happened during the Kapalga Fire Experiment (Williams and Cook 1998, Williams et al. 1999) because of the combination of fuel accumulation to equilibrium levels, and eventual occurrence of human ignition nearby during the late dry-season.

Thus, there is a need for early dry-season burning in savannas, as a tool for both fuel reduction and biodiversity conservation. Application of early dry-season fires in northern Australia is relatively easy, due to predictable fire weather patterns in the savanna biome, and the non-urbanized nature of the landscape. The effectiveness of early dry-season burning in reducing the extent and frequency of late dry-season fires has been demonstrated for the Darwin–Kakadu region (Press 1988, Russell-Smith et al. 1997). Such fuel reduction burning needs to be of relatively low intensity (<1,000 kilowatts per meter) if potential impacts on tree stratum complexity, stem survival, and seed outputs in the tree stratum are to be minimized. An alternative fuel reduction tool may be wet-season burning (Stocker and Sturtz 1966, Williams and Lane 1999), which can reduce fuel loads substantially by reducing the abundance of native species of annual Sorghum. However, consideration also needs to be given to the spatial application of such fires so that those elements of the biota that appear to be sensitive to annual fires are not disadvantaged. Further research is required.

Temperate Eucalypt Woodlands

The problem of fire frequency and biodiversity in the Sydney region reflects the importance of obligate seeder shrub species in most vegetation types. These species are major components of both the flora (hence biodiversity) and the structure (hence fuel) of the woodlands, forests, and overstory of adjacent heaths and shrublands. The key management problem is thus the impact of a high-frequency, low-intensity fire regime on floristic diversity and structural complexity of animal habitats.

Recent studies have verified that this fire regime diminishes floristic diversity (Cary and Morrison 1995, Morrison et al. 1996, Bradstock et al. 1997). Such effects occur when intervals between fires are <5 years. They may become acute when such short intervals occur successively. Moderate fire-return intervals of 10–20 years are typically associated with high levels of floristic diversity and provide a complex vegetative structure. Although experimental evidence suggests that plant species regeneration in these communities is generally a positive function of fire intensity, there is some suggestion that long intervals between fire, e.g., >30 years (Bradstock et al. 1995), may lead to a decline in diversity due to the senescence of both aboveground plants and dormant seed stores in the soil (e.g., Auld 1987). However, this prediction requires more general validation. Areas in such condition are relatively rare due to prevailing fire regimes.

In the Sydney region, fire management for conservation is made considerably more complex than in northern Australia by the existence of an extensive, complex interface between bushland and urban development. Sizable areas of such bushland fall within National Parks, Nature Reserves, Local Government reserves, or other forms of tenure that include conservation as a management objective. Prescribed burning of bushland is perceived as one of the major fire management tools for protecting people and their property in the region. Debate about prescribed burning is ongoing and is usually most vigorous following major fire seasons, such as 1993–1994 and 1997–1998.

Opinion in the community is polarized between those who oppose burning and those who strongly favor extensive, frequent burning. In part such debate reflects confusion about the actual function of prescribed burning in fire management (Gill and Bradstock 1999). Essentially 3 views represent various options for prescribed burning available to managers:

1. The chief role of prescribed burning is to stop either the ignition or the spread of unplanned fires in severe weather in a passive manner, i.e., without the additional input of suppression. The argument is that frequent, extensive applications of prescribed fire are needed to deliver maximum fuel reduction, and hence maximum protection of people and their assets from fire. The amount of protection is assumed to be linearly correlated to the area treated by prescribed burning.

2. Prescribed burning is a tool for moderating intensity of unplanned fires to a level where active suppression is effective (Luke and McArthur 1978). This more selective or strategic approach, involves identifying and treating accessible buffers where prescribed burning can enhance the effectiveness of suppression in difficult conditions.

3. Prescribed burning should focus intensively on the interface between developed land and bushland so that both ember attack on structures (Ramsay et al. 1996) and fire intensity are moderated to manageable and non-threatening levels close to properties.

Bradstock et al. (1998b) investigated the relationship between the extent of prescribed burning and risk of uncontrollable fire at the interface between built assets and bushland. Their model predicts that very high levels of burning (about 40% of the interface annually) are needed to minimize risk of uncontrollable fire (average of 1 day per year) at the interface. A 50% re-
production in burning would lead to a 10-fold increase in risk, given the nature of rapid fuel accumulation, annual occurrence of severe weather, and fairly high-relief terrain typical of the Sydney area. The study indicates that protection levels even within a spatially restricted approach to prescribed burning will be highly sensitive to availability of resources.

Frequent prescribed burning on a broad scale in vegetation communities with a well-developed shrub layer will tend to increase the area subject to frequent fire through an additive effect with unplanned fires. Modelling has been used to gain insight into this problem at the landscape scale. Bradstock et al. (1998a) investigated the use of prescribed burning in simulated landscapes containing vegetation, fuels, and weather typical of the Sydney region. The aim was to understand how passive control of unplanned fires through prescribed burning would affect concurrent protection (size of unplanned fires) and conservation (probability of extinction) objectives.

The level of burning necessary to minimize the size of unplanned fires resulted in high frequencies of fire under the conditions of fuel and weather characteristic of the region (Figure 6). To minimize the size of unplanned fires, about 30% of the landscape needs to be burned each year (Figure 6, Option 1). Such frequencies of fire are predicted to lead to a high risk of extinction of serotinous and leguminous obligate seeders at the landscape scale. Under high frequencies of fire, there is a lower proportion of "points" or "patches" in the landscape with long intervals between fire. The spatial heterogeneity (patchiness) of individual fires may be insufficient to maintain plant species in landscapes when the overall fire regime reaches a critical level. Extinction will not be avoided by patchiness alone of some fires, prescribed or otherwise, in the face of a high-frequency fire regime. Thus, there are critical and non-critical levels of patchiness of individual fires and resultant fire regimes with respect to species' characteristics.

The model can be reparameterized to examine effects of the alternative strategy of using prescribed burning as an active aid to suppressing unplanned fire (Figure 6). Under this option (Figure 6, Option 2), where suppression is simulated as being totally effective in all areas subjected to prescribed burning, a substantial reduction in the size of unplanned fires is indicated. In this option, prescribed burning is more efficient than Option 1, with a substantial reduction in the extent of unplanned fire without high risk of extinction. This result supports the use of selective prescribed burning as a strategic management tool.

Results from modeling exercises such as those described for Figure 6 strengthen the conviction that extensive prescribed burning does not offer the best compromise between protection of humans and biodiversity conservation in temperate eucalypt forests. Thus, several general principles can be derived that define the conundrum of protection versus biodiversity and point toward a strategic solution for temperate plant communities in south-eastern Australia.

1. Prescribed burning to reduce available fuels and the intensity and size of unplanned fires is necessary to protect people and property.
2. Fuel dynamics in conjunction with weather necessitate intervention at a frequency that will reduce biodiversity.
3. Unplanned fires increase overall fire frequency in sites subjected to prescribed fire.
4. The biodiversity problem is potentially "manageable" at the landscape scale provided the number of "points" or "patches" subject to such adverse fire regimes is kept below a critical level. Further research is necessary to determine these levels.
5. High levels of protection of people and property, but not necessarily maximum levels, may be maintained when prescribed fire in the landscape remains below this critical level.
6. Local losses of biodiversity at "points" or in "patches" will result, but landscape-level losses will be avoided if the conditions in Point 5 are achieved.

Despite extremes of debate, actual management has tended to steer a middle course. The use of prescribed fire has generally been strategic, through the creation of buffers designed to enhance suppression effort, often along roads or other fuel breaks, and treatment of the urban interface. This approach has evolved in spite of widespread debate for a number of reasons. First, resources for prescribed burning are limited, which favors smaller rather than large operations. Second, opportunities for burning are limited by appropriate weather and dependence on volunteer labor, which is readily available only on weekends (Gill et al. 1987). Third, the rugged terrain requires intensive use of resources; thus, prescribed burning is expensive. The situation of houses on ridges above narrow canyons leaves little margin for error; therefore, operation
managers are usually conservative in the choice of weather for burning.

CONCLUSION

In the eucalypt forests and woodlands of both south-eastern and northern Australia, the management of biodiversity is closely linked to appropriate fire regimes. In south-eastern Australia the management of biodiversity is linked more closely to fire frequency than to fire intensity. In northern Australia, the management of fire intensity in the landscape is seen as the critical management issue. In both regions, fuel accumulation rates are rapid enough for the landscape to carry fire at short return intervals—annually in northern Australia, and every 2–4 years in the south-east. In both regions, prescribed fuel reduction burning is seen as the primary tool by which unplanned fires may be managed. In south-eastern Australia, the most heavily populated sector of the country, the primary rationale for fuel reduction burning is the protection of life and property. Management in northern Australia more explicitly recognizes biodiversity conservation as a major reason for prescribed burning in the overall management of the landscape. In both regions, however, vulnerable species, both plant and animal, could be at risk from prescribed burning regimes designed to deliver maximum levels of fuel reduction. This risk is particularly apparent in south-eastern Australia, where obligate seeding plants and some small mammals could become locally extinct in the face of a regime of prescribed fires every 3–5 years. In northern Australia, the primary risk to biodiversity appears to be from repeat, high-intensity late dry-season fires. Annual-biennial early dry-season prescribed fires can reduce the incidence of such late dry-season fires. However, some plant and animal species may require fire-free intervals of 2–3 years for effective reproduction and recruitment, and may therefore be at risk from a regime of annual early dry-season fires.

The problem of unplanned, large intense fires during the period of peak fire weather in both regions is largely a contemporary human one, both in terms of protection and origin. It is neither “natural” (large-scale lightning-caused fires are relatively infrequent) nor likely to reflect the nature of ignition rates and their relation to weather before occupation by Europeans. Reducing the rate of human ignitions under severe weather remains part of the solution, with additional benefits such as reducing the attendant risk of fire frequencies adverse to biodiversity conservation. More innovative thinking is needed about how to prevent such ignitions.

No single fire regime is appropriate in either region of Australia for the effective conservation of biodiversity. Finding the balance between area treated and resultant landscape-level fire regimes will continue to be a difficult task in many localities. We stress, however, that prescribed burning is only part of an integrated solution to the problem of human protection from severe fires. A critical area for further research is the integration of point-based studies of fire impacts with a spatial understanding of ignitions, fire behavior, and ecosystem dynamics.

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LITERATURE CITED


Lacey, C.J. 1974. Rhizomes and tropical eucalypts and their role


Williams, R.J. 1995. Tree mortality in relation to fire intensity in a tropical savanna of the Kakadu region, Northern Territory. CALMScience Supplement 4:77–82.


