Chapter 11
Fire and biodiversity

This chapter examines a range of aspects of the impacts of fire on biodiversity in Australia. Two broad types of fires are discussed – wildfires and prescribed fires. Their impacts on plants and animals are discussed. Some species that are vulnerable to the effects of fire and ways to mitigate fire impacts are outlined. Other areas that are briefly explored in this chapter include fire and reserve design, the importance of biological legacies in ecosystem and species recovery after fire, relationships between fire and logging, and the relevance of the intermediate disturbance hypothesis.

11.0 Introduction
Fire has global social, economic, health, and ecological consequences (UNEP, 1999; Gill, 2001; Bradstock et al., 2002). Each year, huge areas of vegetation are burned by wildfires. For example, in 1997–98, the Food and Agriculture Organization (FAO, 2000) estimated that nearly 10 million hectares in Indonesia burned, costing up to US$10 billion and influencing the health of 75 million people. During that time there were major fires in Russia (7 million hectares), Mongolia (2.7 million hectares), Brazil (4 million hectares), and Mexico (800,000 hectares). In Australia, there were major fires in the south-east in 2002 and 2003 (Cary et al., 2003), and extensive areas of northern Australia are burned every year (Commonwealth of Australia, 2001e; Andersen et al., 2003).

Fire influences the majority of Australian terrestrial ecosystems, and many endemic Australian species are threatened by inappropriate fire regimes (State of the Environment, 2001a; Bradstock et al., 2002; Brook et al., 2002b; Keith et al., 2002a). For groups such as birds, only vegetation clearing threatens more species. Attempts to control or prevent high-intensity fire attract public interest; there are conflicts between interests, in part because there is an ongoing expansion of human settlements into areas of flammable and fire-prone native vegetation (Whelan, 2002).

Fires burn differently in different vegetation types, even when they have the same fuel load and are adjacent to one another (Whelan, 1995); apparently similar fires can have very different effects. A fire regime is the sequence of fires typical of a given area. A fire regime has four key components (after Gill, 1975): (1) fire intensity, (2) fire type (e.g., crown or ground fire), (3) between-fire interval (or frequency), and (4) season. Organisms respond to all attributes of a fire, including post-fire conditions such as the amount of precipitation and the features that remain unburned after the fire.

This chapter is limited to a relatively brief appraisal of the relationships between fire and the conservation of biodiversity. Much of this chapter is based on seminal material on fire and biodiversity in Gill et al. (1981, 1999), Whelan (1995), and Bradstock et al. (2002).

11.1 Brief history of fire in Australia
The presence of charcoal in pollen samples collected from swamps around Australia shows that fire has been a prominent part of the Australian landscape for millions of years (Kershaw et al., 2002). Long-term
trends toward a drier climate (since the late Tertiary) coupled with periodic droughts and flammable vegetation probably meant that before the arrival of humans, fire periodically burned large areas of vegetation (Woinarski, 1999). The historical record shows an increase in fire activity about 40 000 years ago, which appears to correspond with increased Aboriginal activity (Kershaw et al., 2002). Fire was used extensively (and still is used) by Aboriginal people (Bowman, 1998, 2003). From an evolutionary perspective, Bowman (2003) argued that: 'one of the great triumphs of the Pleistocene Australians was the taming of wildfires through the development of igniculture'.

Aboriginal people use fire for many reasons:

- to smoke out animals such as possums and gliders from trees (Kerle, 2001; Lindenmayer, 2002b) or to trap smoke-tracking species such as some types of raptors (Boekel, 1980)
- to encourage the development of grasses for herbivores such as Kangaroos (Johnson et al., 1989)
- to stimulate the growth of particular food plants ('firestick farming', sensu Jones, 1969)
- as part of warfare (Martin and Handasyde, 1999)
- to communicate (by signalling; Jones, 1969)
- to clear paths for travel
- to manage fuel loads and landscape mosaics of woodlands and rainforests (Langton, 1998).

Given the widespread use of fire (Bowman, 1998), it is probably not surprising that James Cook referred to Australia as a 'continent of smoke' during his exploration of the east coast in 1770 (Martin and Handasyde, 1999).

There were strong inter-relationships between Aboriginal people, fire, and food supply, as noted by the explorer Thomas Mitchell:

*Fire, grass, kangaroos and human inhabitants, seem all dependent on each other for existence in Australia; for any one of these being wanting, the others could no longer continue* (Mitchell, 1848, p. 412).

Langton (1998) points out that the environmental concerns of Aboriginal people go beyond consumption and the exchange of goods. Fire has spiritual meaning; for instance it is used to ‘clean the country’ after the death of landowners.

Fensham and Fairfax (1996) reported grasslands reverting to rainforest in the absence of Aboriginal burning in the Bunya Mountains of south-eastern Queensland. Regular burning maintained Cypress Pine stands in western New South Wales as open woodlands with an extensive grassy ground cover, but these areas reverted to denser forest when European settlers discouraged burning (Jeans, 1972). Similar changes have been reported for the forests near Sydney (Mitchell, 1848) and in the valley floors of eastern Victoria (Howitt, 1890). Such patterns of fire management and the vegetation cover that resulted might have benefited species that are vulnerable to high-intensity fire (see Box 11.1), but disadvantaged others that prefer habitats with a dense ground cover or understorey.

Bowman (2003) speculated that indigenous fire regimes would have had negative effects on organisms dependent on long-unburned fire refugia, such as several

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**Box 11.1**

**The vulnerability of the Koala to fire**

The Koala is an Australian ‘icon’ species that is well known to all of the nation’s inhabitants and probably the vast majority of people overseas as well. As an arboreal and slow-moving animal that does not use hollows where it can shelter, the Koala is particularly vulnerable to the impacts of wildfires that burn the tree canopy (where the species feeds and roosts). It is notable that even in the early part of the 20th century, leading naturalists such as Fred Lewis, the Chief Inspector of Wildlife in Victoria, regarded wildfire (and not land clearing and shooting for the fur industry) as the key process threatening populations of the Koala (Martin and Handasyde, 1999). It is possible that frequent low-intensity burning of the ground and understorey layers of forests and woodlands by Aboriginal people (e.g. see Howitt, 1890) limited the number of high-intensity conflagrations prior to European settlement. However, populations of Aboriginal people were substantially reduced following European settlement and many of their approaches to land management were no longer practiced. Several extensive wildfires (such as those in 1851, 1898 and 1939) probably contributed significantly to the decline of Koala populations in the forests and woodlands of Victoria. Although prescribed burning does now occur in some parts of the forest estate, frequent major wildfires are still common in eastern Australia (such as those in 1983, 1994 and 2003) and these are likely to harm Koala populations (Martin and Handasyde, 1999).
now-extinct genera of leaf-eating kangaroos (e.g. Stenurus, Simosthenurus and Procoptodon), and probably contributed to their extinction. He also noted that Aboriginal fire regimes maintain some species, for example the Cypress Pine (Callitris intratropica) in northern monsoonal Australia (Bowman, 1998). Irrespective of such speculations, the Australian environment of 200 years ago was the result of millennia of Aboriginal environmental management, and fire was one of their most important land management tools (Bowman, 2003).

Aboriginal fire regimes are tailored to specific locations and conditions to achieve particular objectives (Langton, 1998). In some landscapes, for example the tall montane forests of Victoria, fire does not appear to have been employed routinely. These are cold and wet environments that do not burn readily. In addition, the dense vegetation, tall trees and large quantities of fallen debris characteristic of these areas could have made hunting inefficient relative to the more benign locations nearby.

European settlers, like the Aboriginal people they partially or totally displaced in many parts of Australia, also often used fire in land management, primarily to assist with land clearing. In landscapes such as those on

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**Box 11.2**

**Fires in northern Australia**

In more recent times, there has been increasing attention focused on fire regimes in northern Australia. Organisations such as the Tropical Savannas Management Cooperative Research Centre are researching many aspects of fire impacts on ecosystems and biodiversity. In addition, satellite data are being increasingly used for examining the timing and extent of burning (Bowman et al., 2004; Figure 11.1) – particularly according to Interim Biogeographic Regionalisation for Australia region (as defined in Chapter 4; State of the Environment, 2001a). For example, data from the National Land and Water Resources Audit show that extensive areas of places such as the Kimberley in north-western Australia burn as frequently as three or more times in 7 years (Commonwealth of Australia, 2001e). Moreover, the amount of land burned in northern Australia is increasing annually and in 1998 it was five times greater than in the 4 years previously (25 million hectares versus 5 million hectares; Commonwealth of Australia, 2001e). Other work shows that there are major differences between current fire regimes in areas such as the savanna landscapes and those used by indigenous Australians (State of the Environment, 2001a; Andersen et al., 2005). The timing of these fires are different – fires often now occur much later in the dry season than previously occurred and are extremely intense, which can negatively affect many elements of biodiversity (Pardon et al., 2003), including small mammals (Andersen et al., 2005). The fires that occur now are also non-systematic, extensive and frequent, and result from uncontrolled fires that began elsewhere. This is in marked contrast to the patchy and heterogenous fire regimes used by indigenous people (State of the Environment, 2001a; Bowman et al., 2004). A major management challenge in ecosystems such as the tropical savannas of northern Australia will be to maintain long unburned areas (Andersen et al., 2005).

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**Figure 11.1.** Satellite data showing the extent and frequency of burning in the Kimberley. (Sourced from the Commonwealth of Australia, 2001e.)
Cape Barren Island in Bass Strait, fire was also used to improve hunting. European settlers burnt vegetation to stimulate grass growth and then captured the wallabies that grazed there (Edgecombe, 1999).

Once fences and buildings became features of ‘settled land’, the use of fire by Aboriginal people and European settlers was actively discouraged by the European settlers. Hence, fire regimes in Australia have changed considerably since European settlement (King, 1963; Williams and Gill, 1995; Bowman, 1998; Ward et al., 2001). This is illustrated by the change in the frequency of fires in Snow Gum woodland since 1770 (Banks, 1982; Figure 11.2).

Bowman (2003) speculated that the ‘tamed fire regimes’ developed as part of igniculture then became ‘feral’ following the cessation of Aboriginal fire management. Managing fire is now a challenge for land managers given the extent of infrastructure that now characterises many landscapes, including bushland–urban interfaces (Whelan, 2002). It seems likely that solutions will be found in the approaches developed over thousands of years by indigenous Australians (Bowman, 1998; Whelan, 2003; see Box 11.3). Indigenous fire management varied between vegetation types, landscapes and regions; so, accordingly, there is no single ‘recipe’ for all places (Baker, 2003). Note, however, that reimposing past regimes may not achieve particular conservation goals, for example maintaining rare plant species that have little food or other value for indigenous people (Keith et al., 2002a).

### Box 11.3

**Indigenous fires and better informed fire management in Australia**

Indigenous Australians now manage almost 20% of the continent. Fire was and still is a major management tool used by Aboriginal people (Bowman, 2003; Liddle, 2003). Reintroducing fire into some landscapes by drawing on indigenous knowledge will not prevent wildfires, but it could reduce their impact. However, changes fire management practices need to be made relative to the range of other objectives set for a given area. Moreover, it is also important to note that landscapes are now substantially modified from those managed by indigenous Australians in the past. Factors such as weed invasion can dramatically alter fire regimes and fire impacts on landscapes. Nevertheless, there are important opportunities for exchanges of indigenous and scientific knowledge about fire and land management (Baker, 2003). A potential problem is that although knowledge about indigenous fire regimes is available for many parts of Australia, in other places the knowledge base has been (and continues to be) eroded, thus creating some urgency to keep the remaining information ‘alive’. Land managers can learn many important lessons from indigenous people simply from spending time together in the bush. However, this is related to other key issues, including those of: (1) ensuring that indigenous knowledge is passed on by the ‘right people to other right people’, and (2) how knowledge is stored and who has access to it. There is considerable sensitivity about these issues – Aboriginal people can be protective of their knowledge because in the past it has been exploited by Europeans (Hill, 2003). Part of an approach to address some of these knowledge-management issues has been taken in communities in north-east Australia, where physical examples of fire management practices are applied on the ground to demonstrate to traditional owners and the wider community how ‘country’ can be managed (Davis, 2003).

### 11.2 Types of fire

There are two broad types of fire: **wildfire** and **prescribed** fire. Although they are treated separately in this chapter, they are not always mutually exclusive, as there are many documented cases where prescribed fires have developed into wildfires.
Wildfire
Wildfires are unplanned. They vary substantially in their timing (e.g. the time of the year or time of day), frequency (or return interval; Banks, 1982), intensity (McCarthy et al., 1999a), size (Gill, 1981; Figure 11.3) and heterogeneity (i.e. variation in intensity and impact within the limits of the total area affected; Mackey et al., 2002).

Fire intensity is the amount of heat released at a point on a fire edge, and it is a function of the heat yield of the fuel, the amount of fuel per unit area, and the rate of spread of the fire (Byram, 1959). Gill and Catling (2002) provide a crude scale of intensities: (1) low (<350 kilowatts per metre of fire front), (2) high (350–3500 kilowatts per metre), (3) very high (3500–35 000 kilowatts per metre), and (4) extreme (>35 000 kilowatts per metre). The maximum intensity for burns in Australian forests has been estimated to be 100 000 kilowatts per metre (Gill and Moore, 1994).

Fire timing, frequency, intensity, size and heterogeneity interact. For example, few landscapes experience frequent high-intensity wildfires because fuel loads do not accumulate to high levels. In contrast, many landscapes and vegetation types are characterised by recurrent low-intensity disturbances. Similarly, timing and intensity can be strongly correlated. In the tropical savannas of northern Australia, early dry-season fires are less intense than those late in the same season (Williams et al., 2002). Climate and topography further influence how fire behaves in particular landscapes and vegetation types.

Biological legacies are the structural and ecological elements that remain after fire, including recovering vegetation, fallen and standing logs, ash, soil and canopy-stored seed, and surviving animal populations (Foster et al., 1998; Franklin et al., 2000; see Section 11.10). The numbers, types and spatial arrangements of biological legacies influence the successional dynamics of fire-disturbed plant and animal communities (Lindenmayer and Franklin, 2002; see Section 11.10).

Fires that burn at the wrong time of year can kill plants before seed is set, or interrupt the breeding activities of mammals and invertebrates (Greenslade, 1996; Woinarski and Recher, 1997; Keith et al., 2002b; Whelan et al., 2002). Wildfires that occur too frequently or infrequently can make an area uninhabitable for some species (Gill et al., 1999; Bradstock et al., 2002). We return to the interactions between biota and fire characteristics later in this chapter.

Prescribed fires
Prescribed fires (or ‘hazard reduction burns’) usually involve the routine use of low- to moderate-intensity controlled fires (Morley et al., 2004). These are lit to reduce fuel loads to reduce the risk of extensive, high-intensity fire that can damage property and kill people (P. Cheney, personal communication). Controlling prescribed fires depends on topography, access, wind speed, temperature, humidity, fuel loads and fuel moisture content (Whelan, 1995). Wildfires in recent decades have been suppressed and fuel loads have accumulated in some forest areas to create unacceptable risks to people and infrastructure (e.g. Hurditch and Hurditch, 1994). For example, the Jarrah forests of south-western Australia are burned extensively and frequently to reduce fuel loads (Christensen and Abbott, 1989).
Prescribed fires are sometimes used to manage habitat for wildlife, control weeds (Hodgkinson and Harrington, 1985), and promote plant species for animal survival (as in the case of the Eltham Copper Butterfly; see Box 9.6 in Chapter 9; New, 2000). Prescribed fire is used in montane ash forests and other wet eucalypt forest types in southern Australia. Regeneration burns (or slash burns) are burns that are carried out after timber harvesting to create an ash bed to promote stand regeneration (Campbell, 1984).

Prescribed burning is controversial in Australia. Some vegetation types and the biota they support are susceptible to any fire. Despite the widespread application of prescribed fires in some jurisdictions, in some extreme fire weather situations, prescribed fire may not prevent wildfire. Gill and Bradstock (2003) noted that ‘the extent to which fire regimes can be controlled or imposed by people is largely unknown’.

Frequent burning can select for fire-prone species, making some vegetation types more flammable than they otherwise might have been. In some vegetation types there can be a rapid accumulation of fuel following prescribed fire, so that fuel management can require repeated prescribed fire. For example, Park (1975, in Florence, 1994) notes that:

*Where regrowth develops rapidly following perturbation, the forest floor biomass builds up rapidly to a point of peak fuel energy storage during the forest’s rapid early growth stage.*

Decomposition of the litter layer may make fuel reduction burns unnecessary (Crockford and Richardson, 1998a,b, 2002). Conversely, frequent burning can destroy the organisms that decompose litter, resulting in a rapid accumulation of new litter requiring further burning.

Equilibria between decomposition and accumulation are not characteristic of all fuel types in all vegetation types. In the Mountain Ash forests of Victoria, the volume of large logs increases with stand age (Lindenmayer et al., 1999a). However, bryophyte cover on logs, log decay levels, and the moisture content of logs also increases with age (Ashton, 1986; Lindenmayer et al., 1999a). There are no simple, linear relationships between fuel accumulation, fuel levels, fuel flammability and time since fire.

Prescribed burning is expensive and can compete for resources with other management activities. The costs of maintaining a prescribed burning program over large areas of the national park and State forest estates would be prohibitive, even if it was feasible (Whelan, 2002).

Prescribed fire involves other trade-offs. Perhaps most importantly, fire affects the amount and quality of water produced by catchments. Smoke can have impacts on natural assets, such as discoloring cave formations (A. Spate, personal communication). Smoke can influence air quality and can pose a human health risk, particularly for people with respiratory conditions such as asthma (Johnston et al., 2002).

Some areas do not burn readily, making management of fuel with prescribed fires difficult. In the montane ash forests of the Central Highlands of Victoria, for example, the high moisture content of the large quantities of woody debris on the forest floor (see Ashton, 1986) make low-intensity prescribed fires almost impossible. Thus, most wildfires in landscapes dominated by these forest types are high-intensity ones (Smith and Woodgate, 1985; Attiwill, 1994a,b; McCarthy and Lindenmayer, 1998). If forest biotas are adapted to infrequent, high-intensity fires, low-intensity fuel-reduction fires can produce unwanted ecological outcomes (Lindenmayer and Franklin, 2002). Repeated low-intensity prescribed fire can affect the biota negatively or positively (Gill et al., 1999). We explore this topic in the following section.

### 11.3 Response of biodiversity to wildfire

#### Wildfire and Australian animals

Fires can kill many individual animals at the time of the conflagration (e.g. birds in the Nadgee Nature Reserve...
on the far south coast of New South Wales in 1972; Fox, 1978; Keith et al., 2002b; see Table 11.1). Survival of individual animals depends on mobility (to escape fire) and the insulating potential of fur and feathers (Whelan, 1995). Additionally, many animals that survive fire subsequently perish because of limited food and shelter, or because of increased predation due to a lack of vegetation cover (Christensen, 1980; Russell et al., 2003).

Species have a wide array of responses to a single fire (Letnic, 2003). There can be different responses within a given vegetation type, between vegetation types, and following successive fires at the same site (Fox, 1982; reviewed by Whelan, 1995; Woinarski, 1999; Whelan et al., 2002). Figure 11.4 shows the changes in occurrence or abundance with time since fire of a small subset of birds and small mammals. Some species are attracted to burning areas and others, for example some raptors, can even attempt to promote the spread of fire by dropping smouldering sticks in unburned areas. The Black Kite and the Pied Butcherbird find prey more easily in burned areas (Woinarski and Recher, 1997). Fire-damaged areas provide suitable habitat for some species, for example several species of beetles (Schmitz and Bleckmann, 1998), including jewel beetles (Beattie and Ehrlich, 2001). A range of invertebrates forage on burned logs and fire-damaged standing trees (Grove et al., 2002).

Wildfires promote habitat development for some animal species after a given interval following disturbance (Catling et al., 2001) through stimulating the germination and subsequent growth of particular food plants (e.g. Meredith et al., 1984). Wildfires can produce significant quantities of dead wood (including logs) and stimulate the development of cavities in trees (Inions et al., 1989), which are essential habitat elements for many animal species, and which can be depleted by forestry operations (Grove, 2001; Gibbons and Lindenmayer, 2002) and prolonged high-intensity grazing by domestic livestock (Gibbons and Boak, 2002).

Wildfire suppression can also affect the biota. Woinarski et al. (2004) found a range of species that were strongly associated with long unburned areas in tropical open forests in northern Australia. In other cases, back-burning is used to check the spread of wildfires, but it can close off the escape options for animals and incinerate them (Baker, 1997). Roads built to give access to fire-fighters can cause damage, providing conduits for the spread of weeds (Wace, 1977), causing soil erosion and stream sedimentation (OShaughnessy and Jayasuriya, 1991), and facilitating the movement of feral predators (May and Norton, 1996). After wildfires in Victoria in 2003, aerial mapping showed that several thousand kilometres of roads had been constructed to fight the fires.

Fire-damaged forests are sometimes logged (termed salvage logging). Post-fire salvage logging in Victorian forests following the 1939 wildfires continued for nearly 20 years (Noble, 1977). The logging significantly reduced the abundance of large trees with hollows and probably had corresponding negative effects on an

Table 11.1. Numbers of birds killed directly in Nadgee Nature Reserve after the 1972 wildlife. (Modified from Fox, 1978; Keith et al., 2002b.)

<table>
<thead>
<tr>
<th>Species</th>
<th>No. carcasses</th>
</tr>
</thead>
<tbody>
<tr>
<td>New Holland Honeyeater</td>
<td>266</td>
</tr>
<tr>
<td>Little Wattlebird</td>
<td>120</td>
</tr>
<tr>
<td>Eastern Yellow Robin</td>
<td>49</td>
</tr>
<tr>
<td>Brown Thornbill</td>
<td>25</td>
</tr>
<tr>
<td>Rufous Whistler</td>
<td>21</td>
</tr>
<tr>
<td>Eastern Spinebill</td>
<td>12</td>
</tr>
<tr>
<td>Striated Thornbill</td>
<td>11</td>
</tr>
<tr>
<td>White-throated Treecreeper</td>
<td>10</td>
</tr>
<tr>
<td>Crimson Rosella</td>
<td>7</td>
</tr>
<tr>
<td>Blackbird</td>
<td>5</td>
</tr>
<tr>
<td>Ground Parrot</td>
<td>5</td>
</tr>
<tr>
<td>Yellow-faced Honeyeater</td>
<td>5</td>
</tr>
<tr>
<td>Brush Bronzewing</td>
<td>4</td>
</tr>
<tr>
<td>Superb Fairy Wren</td>
<td>3</td>
</tr>
<tr>
<td>Beautiful Firetail</td>
<td>3</td>
</tr>
<tr>
<td>Red-browed Finch</td>
<td>3</td>
</tr>
<tr>
<td>Crested Shrike Tit</td>
<td>2</td>
</tr>
<tr>
<td>Grey Butcherbird</td>
<td>2</td>
</tr>
<tr>
<td>Owlet Nightjar</td>
<td>2</td>
</tr>
<tr>
<td>Pied Currawong</td>
<td>2</td>
</tr>
<tr>
<td>Stubble Quail</td>
<td>2</td>
</tr>
<tr>
<td>Varied Sitella</td>
<td>2</td>
</tr>
<tr>
<td>Bell Miner</td>
<td>1</td>
</tr>
<tr>
<td>Boobook Owl</td>
<td>1</td>
</tr>
<tr>
<td>Barn Owl</td>
<td>1</td>
</tr>
<tr>
<td>Eastern Whipbird</td>
<td>1</td>
</tr>
<tr>
<td>Fan-tailed Cuckoo</td>
<td>1</td>
</tr>
<tr>
<td>Grey Shrike Thrush</td>
<td>1</td>
</tr>
<tr>
<td>King Parrot</td>
<td>1</td>
</tr>
<tr>
<td>Leaden Flycatcher</td>
<td>1</td>
</tr>
<tr>
<td>Olive-backed Oriole</td>
<td>1</td>
</tr>
<tr>
<td>Rainbow Lorikeet</td>
<td>1</td>
</tr>
<tr>
<td>Satin Bowerbird</td>
<td>1</td>
</tr>
<tr>
<td>Scarlet Robin</td>
<td>1</td>
</tr>
<tr>
<td>Southern Emu Wren</td>
<td>1</td>
</tr>
<tr>
<td>Spotted Pardalote</td>
<td>1</td>
</tr>
<tr>
<td>Tawny Frogmouth</td>
<td>1</td>
</tr>
<tr>
<td>White-browed Scrub Wren</td>
<td>1</td>
</tr>
<tr>
<td>White-naped Honeyeater</td>
<td>1</td>
</tr>
</tbody>
</table>

Wildfires promote habitat development for some animal species after a given interval following disturbance (Catling et al., 2001) through stimulating the germination and subsequent growth of particular food plants (e.g. Meredith et al., 1984). Wildfires can produce significant quantities of dead wood (including logs) and stimulate the development of cavities in trees (Inions et al., 1989), which are essential habitat elements for many animal species, and which can be depleted by forestry operations (Grove, 2001; Gibbons and Lindenmayer, 2002) and prolonged high-intensity grazing by domestic livestock (Gibbons and Boak, 2002).
array of cavity-dependent species (Lindenmayer et al., 2004b). Salvage logging also has detrimental impacts on biota in many other forest ecosystems (e.g. Hutto, 1995; Foster et al., 1997; Saab and Dudley, 1998; McIver and Starr, 2001).

**Box 11.4**

**The impact of salvage harvesting in forests**

Many ecologists view natural disturbances such as wildfires as key ecosystem processes rather than ecological disasters that require human repair (e.g. Turner et al., 2003). Indeed, major disturbances can sometimes aid ecosystem restoration by recreating some of the structural complexity and landscape heterogeneity lost through previous intense management (e.g. intensive forestry). For example, major wildfires can generate significant volumes of dead trees and logs that provide important habitat for many organisms, but which are depleted by traditional forestry practices. However, salvage harvesting operations in forests can have significant negative impacts on ecosystems and undermine many of the benefits of major disturbances. The negative ecological impacts of salvage operations can manifest in at least three ways.

First, there can be major impacts on ecosystem processes following salvage operations. Extensive salvage harvesting after the 1938 New England hurricane produced a long-lasting shift in hydrological regimes that were manifested at a regional scale (Foster et al., 1997). Second, the removal of large quantities of biological legacies threatens the persistence of some taxa. For example, salvage harvesting of burned trees removes critical habitat for many early successional species (e.g. Hutto, 1995). Third, there can be compounding, cumulative or magnified effects on ecosystem processes and elements of the biota if an intense natural disturbance event is soon followed by an intensive (and often prolonged) human disturbance. That is, some taxa may be maladapted to the interactive effects of two disturbance events in rapid succession. For example, in South-East Asia, salvage logging of burned rainforests led to significant forest deterioration and loss with major negative impacts on the regenerative potential of stands, as well as a range of other undesirable effects (van Niewenstadt et al., 2001). The impacts of salvage harvesting indicate a need for large areas to be exempt from such activities, including national parks. In addition, carefully formulated policies are needed wherever salvage harvesting continues in order to ensure the retention of biological legacies such as dead trees, live trees, logs, and islands of undisturbed or partially disturbed vegetation (Lindenmayer et al., 2004b).
Some plants survive fire and then resprout from epicormic buds, roots or lignotubers. Others are killed by fire, but germinate from fire-resistant seeds in the canopy or soil. The seeds of some species (e.g. various species of Acacia) remain viable for many hundreds of years.

The timing of fire can be critical. Dry season burning in northern Australia can have no apparent negative effects on some grass species, but a wet season fire will eliminate populations because there is no seed bank at that time of year (Stocker and Onwin, 1986).

Fire frequency can also have a major impact. In the case of Mountain Ash forest in Victoria, trees do not reach sexual maturity until they are at least 20 years old.
and seeds do not remain viable in the soil for a prolonged period like those of many other plant species. Stand-replacing fires at intervals shorter than 20 years (e.g. those in 1926 and 1939) lead to replacement by stands of Acacia spp. trees. Reinvasion of Mountain Ash trees then needs to take place from seeds dispersed from adjacent stands (Mackey et al., 2002). If the fire interval exceeds the lifespan of Mountain Ash (>350–500 years), then eucalypt stands will eventually be replaced by Myrtle Beech in parts of landscapes suitable for the development of cool temperate rainforest (see Lindenmayer et al., 2000a,b).

There can be large differences in the sensitivity of individual plants at different times in their life cycle. For example, large old eucalypts can be more fire resistant (because of thick bark) than younger seedlings or saplings (West, 1979). Conversely, very large and decayed trees can be very fire-prone because of the accumulation of flammable rotting material and the limited amount of water in the conducting tissues of the stem (Gibbons and Lindenmayer, 2002).

Plants respond in different ways to fire: some species grow more rapidly after fire, others undergo massive flowering (e.g. Grass Trees), and yet others germinate seedlings en masse (e.g. Mountain Ash and Alpine Ash). Responses are influenced by variables such as (among others) post-fire climate conditions (Whelan, 1995), the life history attributes of the species (Noble and Slatyer, 1980), and the quality (productivity) of the sites (Ashton and Martin, 1996).

Wildfire and identifying patterns of species responses

The array of species responses to fire gives rise to what Bowman (2003) termed a ‘blizzard of ecological details’. In an effort to simplify the problem, several authors (e.g. Whelan, 1995; Gill, 1999) have developed response curves to explain changes in biota with time since fire using measures such as species diversity, abundances of particular taxa, and rank abundance (of plants). Some examples are shown in Figure 11.5, and they highlight the array of possible species responses to fire: some species remain relatively immune or

![Image of Grass Trees on the floor of a coastal woodland two weeks after a fire, that are beginning to resprout. (Photo by David Lindenmayer.)](Image)

**Figure 11.5.** (A) Hypothetical plant species diversity response curves with time since fire (after Gill, 1999). (B–F) Hypothetical individual population size response curves with time since fire (redrawn from Whelan et al., 2002).
benefit considerably from repeated fire, whereas others are impacted negatively.

General fire response patterns, such as those based on particular life history attributes (e.g. animal dispersal ability, or whether a plant is a resprouting or an obligate seeding plant) can assist managers to predict the responses of species assemblages to particular management regimes (such as fire exclusion or frequent prescribed fire). Noble and Slatyer (1980) proposed a model for plant responses based on ‘vital attributes’. Three groups of attributes were proposed in their model:

- method of arrival and/or persistence of a given species during and following disturbance
- establishment and growth of a plant species in a plant community
- time required for the species to reach key life history stages (e.g. to produce viable seed).

A given vital attribute can occur as a result of different biological mechanisms (Noble and Slatyer, 1980).

Response categories are only approximate guides to ecological responses (Whelan, 1995; Whelan et al., 2002). For example, a species may or may not resprout, depending on fire intensity. Myrtle Beech can resprout after low-intensity fires, but is killed outright by high-intensity conflagrations. Whelan et al. (2002) concluded that for animals, attempts to identify ‘clear associations between fire responses and fire regimes’ have not been particularly fruitful. Outcomes are often site-specific, and there is unlikely to be a set of ‘vital attributes’ that determines responses for animals as there is for plants. In fact, fire responses vary within and among sites even for the same species (e.g. the Ground Parrot discussed in Section 11.7; Woinarski, 1999; Keith et al., 2000b). Whelan et al. (2002) argued that more (and better designed) experiments are required to identify the underlying ecological processes that give rise to the emergent fire-response patterns.

A fire regime that benefits a particular species or set of species may not suit others. Keith et al. (2002a) noted that ‘the full spectrum of biodiversity cannot be maintained if fire management only addresses a single species in isolation’. This injects considerable complexity into the inter-related topics of wildfire, fire management and biodiversity conservation. The objectives of management need to be carefully articulated (as examined later in this chapter).

11.4 Response of biodiversity to prescribed fire

The impact of prescribed burning on biodiversity is controversial and research results are equivocal (reviewed by Williams and Gill, 1995; York, 1999; see also Collett, 2003 among many other studies).

King (1985) found that log cover was reduced by prescribed burning and concluded that there could be cumulative impacts on litter invertebrates and their small mammal predators. Similarly, in a study by State Forests of New South Wales Northern Region (1996) it is noted that frequent low-intensity prescribed fires accelerated the decay of large logs. Smith et al. (1992) and Hannah et al. (1998) found that both logs and many vertebrates were less common in frequently burned forests (see also Singh et al., 2002). In one of the longest running studies completed to date, York (1996, in Gill et al., 1999) showed how prescribed burning changed the extent of charring and desiccation of the outer surfaces of logs, leading to a change in the composition of invertebrate communities to include more species that were more tolerant of drier and more open forest environments (York, 1999, 2000).

Fire is used on farms to reduce fire hazards around paddocks and farm buildings. As in the case of forests and woodlands, such practices can affect biota that are dependent on logs, fallen timber and leaf litter. Lindenmayer et al. (2003a) recommended that ecologically-sensitive areas on farms such as wetlands (especially those with peat) be quarantined from fire; elsewhere, burning should be concentrated on paddock boundaries, leaving the interior parts unburned or less intensively burned.

Burning off in a low-lying swampy area – an activity that can cause long-term damage to these habitats. (Photo by David Lindenmayer.)
Mosaic or patchy burning can maintain suitable habitat for some ground-dwelling organisms and provide escape routes for others. The interval between burning should be sufficient to regenerate native vegetation and allow fire-sensitive species to recover. For example, Lambert and Elix (undated) and McIntyre et al. (2002) recommended that the intervals between fires in grassy woodlands should be at least 5–10 years. Fire can facilitate weed establishment and promote weed growth (Hobbs and Atkins, 1991), especially if fires are frequent (Hobbs and Atkins, 1990).

Prescribed burning is designed to reduce fuel loads and, in turn, reduce the risk of a high-intensity fire. In some ecosystems, frequent prescribed fires will be necessary to reduce fuel. However, fuel is also habitat for animals and plants (e.g. logs that provide nursery sites for plant germination; Howard, 1973; McKenny and Kirkpatrick; 1999); therefore, changes to fuel such as logs and leaf litter can impact on the elements of the biota that depend on these features (York, 1999; Lindenmayer et al., 2002b).

11.5 Species vulnerability to fire

Animal and plant groups threatened by altered fire regimes

Inappropriate fire regimes threaten biodiversity in Australia (State of the Environment, 2001a). In fact, they have already contributed to the extinction of two species and three subspecies of Australian birds (Woinarski, 1999). For example, the Kangaroo Island Emu – a dwarf species in comparison to the mainland form – is thought to have become extinct because of wildfires in the 19th century (Ford, 1979). Inappropriate fire regimes threaten more than 50 extant bird species, they are second only to land clearing as a threat to avifauna in Australia (Garnett, 1992; Maxwell et al. 1996; Australian Bureau of Statistics, 1999b). For example, the Malleefowl is most abundant in long-unburned mallee habitats (Bradstock and Cohn, 2002), probably because of its reliance on litter for nest building (although it is also vulnerable to other factors, such as predation by the Red Fox).

Approximately 45% of south-western Australia’s mammals and birds may be negatively influenced by altered fire regimes (Calver and Dell, 1998) (e.g. the White-bellied and Orange-bellied Frogs; Wardell-Johnson and Roberts, 1993).

Leigh and Briggs (1992) list almost 20 plant species that are threatened with extinction because of inappropriate fire regimes. There are also many examples of local plant extinctions (Leigh et al., 1984; Gill and Bradstock, 1995) and changes in the composition of vegetation communities as a result of inappropriate fire regimes (e.g. Niewenhuis, 1987). The decline of fire-sensitive Mulga stands in arid Australia may be due to infrequent large fires in spinifex ecosystems (Gill, 2000). Previously, these communities experienced more frequent, smaller fires as part of Aboriginal management (Allan and Southgate, 2002).

Vegetation communities sensitive to fire

Some vegetation communities in the Australian environment appear to be particularly sensitive to the impacts of fire, suggesting that they have evolved largely in the absence of fire. An endemic Tasmanian conifer,
King Billy Pine, is highly sensitive to fire (Brown, 1988). Living trees of this species can exceed 1000 years of age. Stands of the species in western Tasmania were damaged by a fire in 1914, and there has been only minimal regeneration since then.

Some wetland habitats, such as those that contain peat, require protective management because they are readily and badly damaged by fire, and take a long time to recover. Fires in peat bogs can dramatically reduce the quality of water within adjacent creeks and destroy drought refuge areas for native animals (Lindenmayer et al., 2003a). Fire can damage moist high-elevation vegetation types such as endemic plant-rich feldmark (which often supports long-lasting snow patches) and sphagnum bogs. Species that are closely associated with sphagnum bogs, for example the Corroboree Frog, may be particularly sensitive to wildfires (K. Green, personal communication).

Rainforest is also sensitive to fire (Bowman, 1999; Fensham et al., 2003). In the prolonged absence of fire, cool-temperate rainforests (e.g. those in Tasmania; Jackson, 1968), warm-temperate rainforests, tropical rainforests (Harrington and Sanderson, 1994; Mullen, 1995), and monsoonal rainforests (Russell-Smith and Bowman, 1992) invade adjacent vegetation (such as wet sclerophyll forest).

Wet sclerophyll forests in northern Queensland occur as long narrow belts adjacent to tropical rainforest that is protected from logging and fire (Figure 11.6). Wet sclerophyll forest supports many species that are found virtually nowhere else, for example the northern subspecies of the Yellow-bellied Glider and the Northern Bettong (Laurance, 1997a,b; see Table 11.2). Gill (1999)
describes a model of how sclerophyll forest can be invaded by rainforest (see Figure 11.7), a process that could have a significant impact on the persistence of many species (Table 11.2). The conservation of biodiversity closely allied with wet sclerophyll forests together with that of adjacent rainforest will require management of different disturbance regimes in close proximity.

### 11.6 Spatial variability in fire behaviour: fire refugia, landscape mosaics, and Aboriginal burning patterns

Wildfires and prescribed fires rarely burn all areas with equal intensity, and this spatial variability can be important. For example, patches that escape fire provide refugia for fire-sensitive taxa (Mackey et al., 2002). Animals and plants survive fire in microhabitat features such as rock piles, soil cracks and tree hollows (Clarke, 2002; Whelan et al., 2002). Large logs can be fire refuges for plants and animals because of their diameter and length, the moisture they contain, and the moisture levels in the adjacent litter (Campbell and Tanton, 1981; Andrew et al., 2000; reviewed by Lindenmayer et al., 2002b). For example, Christensen (unpublished data) in Christensen et al. (1981) recorded large numbers of skinks surviving under logs after moderate- to high-intensity wildfires.

At a landscape scale, streams, lakes and rocky cliff-lines are natural firebreaks. Differences in flammability between different vegetation types mean that, for example, sedgelands burn more readily than neighbouring woodlands, thus reinforcing heterogeneity in landscape vegetation patterns (Burrough et al., 1977). Within widespread areas of the same vegetation type, microclimate, terrain and vegetation structure create fire refugia. In the ash-type forests in the Central Highlands of Victoria, steep slopes and areas with low levels of radiation experience lower fire intensities. In these areas, more trees survive and multi-aged stands are relatively common (Mackey et al., 2002). Multi-aged forests support the highest diversity of species of arboreal marsupials (McCarthy and Lindenmayer, 1998).

### Fine-scale vegetation mosaics and Aboriginal burning

Aboriginal people burned the spinifex-dominated landscapes in dry inland parts of Australia, producing a fine-scale vegetation mosaic that provided suitable habitat for medium-sized native mammals (Bolton and Latz, 1978; Burbidge et al., 1988). These landscapes now experience large fires (Burrows and Christensen, 1990) and suitable habitat for medium-sized native mammals may have been lost (see Short and Turner, 1994; Allan and Southgate, 2002).

In the examples of the montane ash forest and the spinifex landscapes, fire patterns and the resulting heterogeneity in vegetation at local and landscape scales have important implications for biodiversity (Lindenmayer and Franklin, 2002; Allan and Southgate, 2002). A range of structural and floristic conditions support more taxa than one or only a few vegetation conditions. Fire management is an important part of conservation management, which is the topic of the following section.
11.7 Fire management and biodiversity conservation

The most appropriate fire regime for a system will depend on the objectives of management and the characteristics of the system. Management may involve both the suppression of unwanted fires (including wildfires) and the ignition of prescribed fires (Gill, 1999). Objectives for management will vary depending on the proximity of people and property and the relative importance of conservation and water production. The situation is complicated by the fact that few (if any) areas have just one economic or ecological value (Keith et al., 2002a).

Even if biodiversity conservation is the primary management objective, differences between vegetation communities and individual elements of the biota in their response to fire means that there are no simple recipes. Given such complexity, what can be done? One approach is to vary fire regimes between and within landscapes, creating a range of conditions. Therefore, if unsuitable habitats are created in one area, there will be other places where a species can survive. This is termed ‘risk-spreading’ (den Boer, 1981; see also Lindenmayer and Franklin, 2002). Large parts of some landscapes, for example the edges of national parks bordering grazing properties or urban developments, might be subject to frequent low-intensity prescribed burns (as is now done by the New South Wales Department of Environment and Conservation; Whelan, 2002). Other landscapes that are more remote from human infrastructure may be burned less frequently and/or less of the area may be burned, and others still may remain unburned by prescribed fire (although wildfire may still occur).

There are two key variables in this simple example: the interval between prescribed fires (fire frequency) and the proportion of a landscape burned (fire extent). Unfortunately, very little is known about how frequent and extensive prescribed fire must be to reduce the risk of wildfire (see Cary and Bradstock, 2003), or about the impacts on biodiversity of varying prescribed fire frequencies and sizes. Such uncertainty highlights the need to record actions, monitor the response of the biota, then feed these observations back to managers so that practices could be improved. This process is called adaptive management or learning by experiment and monitoring (Walters and Holling, 1990; Bunnell et al., 2003; see Chapter 19). Despite its potential to improve biodiversity outcomes, the method has rarely been used by resource management agencies (Dovers and Lindenmayer, 1997; Beese et al., 2003; Bunnell et al., 2003).

The prescribed burning example given in this section is highly simplified. Landscapes are not homogeneous — some areas, such as the wet gullies in valley floors, are less likely to burn than other places. Conducting patchy burns of varying intensity is not easy in practice, and the timing and intensity of burning add further complexity. Such variation may provide opportunities for natural experiments (see Box 10.11) and a feedback process to inform management.

In summary, perhaps the best informed perspective on fire management comes from Gill (1999, p. 47), who believes that the management of fire for biodiversity conservation should:

aim at achieving suitable proportions of landscape with a variety of times-since-fire stages within appropriate intensity levels at appropriate times of the year and within appropriate frequency range.

This is an important objective, but it is not easy to achieve, because using fire as a management tool is influenced by many practical constraints, including in many cases a limited time window during which prescribed fires can be applied safely (Whelan, 1995), and limited financial and human resources for fire management. Where prescribed burning is difficult, it may be better to use other forms of management (Baker, 1997). For threatened grassland lizards such as the Striped Legless Lizard and the Grassland Earless Dragon, low-intensity grazing may be more appropriate than frequent prescribed burning (Robertson and Cooper, 1997; Robertson, 1999).

Fire management and conservation of the Eastern Bristlebird and the Ground Parrot

Active management is sometimes required to maintain or restore important ecological values or conserve particular taxa. Two high-profile cases are the Eastern Bristlebird and Ground Parrot, which are birds that are associated with heathlands (Garnett, 1992). In general, field data suggest that many areas of heathland have been burned more frequently than is desirable for some threatened bird species (Woinarski, 1999).

The Ground Parrot is a strong flier that quickly recolonises burned areas and then occupies them for many years (see Woinarski, 1999; Keith et al., 2002b). Early work suggested that burning at intervals of 4–5 years was appropriate for this species. More recently,
recommendations have extended the time between fires to 8–10 years in Queensland and south-west Western Australia and 10–25 years in Victoria (Woinarski, 1999). In Victoria, animals were least common in heaths that were less than 3 and more than 18 years old (Meredith et al., 1984). However, Baker and Whelan (1994) found that Ground Parrot populations did not decline with heathland age and recommended fire exclusion. The most appropriate fire management strategy for the Ground Parrot may vary between areas and in relation to food resources such as seed production in particular habitats.

The Eastern Bristlebird requires multi-layered heathland vegetation (Baker, 1998). It disperses poorly and appears to be vulnerable to the effects of large-scale fires that do not leave unburned refugia (Pyke et al., 1995; Baker, 1997). Fire exclusion is the most appropriate conservation strategy, but this may not be possible given the proximity of human infrastructure to its habitat, such as at Jervis Bay Village on the south coast of New South Wales. Because of the need for fuel management in these areas, Baker (1997) recommended the strategic slashing of vegetation. In other cases where prescribed burning is essential, the direction of fire fronts should be planned to provide escape routes for animals,
given their limited movement ability. Many of these recommendations have been embraced as part of management of Eastern Bristlebird habitat, such as that in the Booderee National Park in New South Wales (M. Fortescue, personal communication). Nevertheless, in late 2003, a major wildfire burned much of the complex multi-layered heathland habitat occupied by the Eastern Bristlebird.

The Eastern Bristlebird and the Ground Parrot co-occur in some areas of heathland, including those that may be periodically burned. Baker (1997) believed that the management of the Eastern Bristlebird should have priority in these circumstances (primarily achieved through fire exclusion) because its distribution is more restricted, it has poorer recolonisation ability, and its populations are smaller and more sensitive to frequent fire. This situation emphasises the importance of setting explicit objectives for ecological management, and in so doing making assessments of the potential trade-offs involved.

### 11.8 Studies to examine the effects of fire

One of the consistent themes emerging from the literature on fire is reinforced by statements in the most recent State of the Environment Report (2001, p. 8): 'the effects of various fire regimes on the conservation of biodiversity remain uncertain'. There is general consensus about the need for more detailed work on the impacts of fire on the Australian biota. But what are the best ways to study fire? There are several ways improving our knowledge of fire impacts, including experiments, observational studies and modelling. Each is briefly outlined below.

#### Experiments

Fire experiments are critical because they identify the underlying ecological processes that give rise to patterns of species response (Whelan et al., 2002). However, the experimental design of many fire studies is poor, limiting the validity of inferences. For example, many studies take place only after a fire has occurred (Whelan, 1995). A key part of experiments is that many factors need to be controlled to test a small number of variables of interest. Experiments can enable robust conclusions to be reached about, for example, the effects of the timing of a fire on small mammal population dynamics if conducted with adequate replication of treatments, quantification of before-treatment conditions in each replicate, and replicated ‘control’ sites that are matched to the treatments (see Whelan, 1995).

The interpretation of fire effects from experiments (and other types of studies) needs to be made in

<table>
<thead>
<tr>
<th>Record of dead organisms after fire</th>
<th>Indication of population change</th>
<th>Separation of fire effects from temporal change</th>
<th>Separation of fire effects from site effects</th>
<th>Inference of fire effect on population</th>
</tr>
</thead>
<tbody>
<tr>
<td>Census before and census after fire</td>
<td>Yes, but choice of time for post-fire sample may mean a change is missed</td>
<td>No</td>
<td>No</td>
<td>No</td>
</tr>
<tr>
<td>Census before and after fire in a burned and a control site</td>
<td>Yes</td>
<td>Yes, but there will be a confounding interaction of season and fire effects if timing of samples not carefully planned</td>
<td>No</td>
<td>No</td>
</tr>
<tr>
<td>Census before and after fires in randomly allocated replicate burned and control sites</td>
<td>Yes</td>
<td>Yes</td>
<td>Yes</td>
<td>Yes, but care must be taken not to treat consecutive samples as replicates, as they are not independent</td>
</tr>
</tbody>
</table>
conjunction with an understanding of the biology of the species in question. For example, if long-lived species survive after fire, their persistence may mask impacts on other processes such as reproduction. In other cases, more captures of animals after fire may be due to increased movements to find food rather than greater abundance (Whelan, 1995). In the case of plants, many taxa that appear after fire were present before the fire in cryptic forms such as dormant seeds or ground tubers (Gill, 1999).

Despite the considerable value of experiments, it is surprising how few have been conducted in Australia (and elsewhere around the world; Whelan, 1995). Studies such as the Kapalga Fire Experiment in Kakadu National Park in the Northern Territory (see Braithwaite, 1995; Brook et al., 2002b; Andersen et al., 2005 among numerous other publications) are among the few Australian exceptions. Perhaps part of the reason for the lack of experiments is that experimental fires have to be of low to moderate intensity and constrained to small areas, otherwise they can become uncontrolled wildfires. Hence, they cannot represent the wide range of intensities that occur in nature (Gill, 1977).

Because experiments often take place on a small spatial scale, they can be of limited value for examining species that move over larger scales, such as birds and large mammals. By carefully controlling some conditions in experiments, initially unforeseen biases can be added, which confound the results and the interpretation of effects (Huston, 1997). Complex interacting factors that can have an important influence on species persistence, such as cumulative effects (sensu McComb et al., 1991; Burris and Canter, 1997) in which two factors in isolation have limited impacts but are substantial in combination, often cannot be readily examined with field experiments. Finally, experiments can be expensive to establish and maintain, but changes in populations (or other measures of interest) can take a long time to appear.

**Observational studies**

The vast majority of studies of fire in Australia have been observational investigations that took advantage of unplanned fires. Typically, they lack the replication of sites and the controls that characterise well-designed experiments (e.g. Thompson et al., 1989; Loyn, 1997; Singh et al., 2002; see Table 11.3). Despite the inherent problems of observational studies, they can nevertheless produce important insights into the impacts of fires on biota (Gill et al., 1999), including the effects of very high-intensity fires (e.g. Recher et al., 1975, 1985). Indeed, much of what is currently known about fire impacts on biota comes from observational studies (reviewed by Whelan, 1995; Woinarski, 1999).

**Modelling**

The value of modelling in studies of fire is controversial, with some authors arguing that modelling has provided few valuable insights, whereas others contend the opposite (see debates in Cary et al., 2003). However, we believe that modelling can make a useful contribution to a better understanding of fire and its impacts on the biota, particularly when it is underpinned by sound ecological data (Burgman et al., 1993).

Modelling can be a vehicle for synthesising what is known about particular phenomena (Burgman et al., 1993). It can help form hypotheses for testing with field-based empirical studies, such as in adaptive management programs (e.g. Walters, 1986). It can provide a mechanism to compare the relative effectiveness of management options (e.g. the risk of extinction of a given species under several conservation strategies; Possingham et al., 1993). With modelling, it is possible to describe the possible long-term consequences of actions taken now. For example, McCarthy and Burgman (1995) demonstrated that fire and logging led to quite different amounts of forest in different age classes in a landscape, even if the average fire-return interval was the same as the logging rotation time (Figure 11.8). This has implications for forest management and the conservation of species associated with
particular age classes, and would not have been identified from long-term field observations. Similarly, phenomena such as the sensitivity of fire regimes to future climate change cannot be studied empirically (Cary, 2002). Despite the value of modelling, model-users should be clear about the limitations of models.

Another important role for modelling is that of risk assessment, in which different management options can be compared and ranked in terms of the likelihood of an unfavourable conservation outcome (e.g. species extinction). Such management options can include the application of fire-related actions or possible chance occurrences of wildfires. There are some useful examples of such applications of risk assessment modelling to the conservation of plants and animals, including the Heath Banksia (Bradstock et al., 1997), Splendid Fairy Wren (Brooker and Brooker, 1994), Carpentarian Rock Rat (Brook et al., 2002b), and Leadbeater’s Possum (Lindenmayer and Possingham, 1995; McCarthy and Lindenmayer, 2000). Richards et al. (1999) used a decision theory framework to explore fire management options within reserves (Box 11.7).

In summary, all three types of investigations – experiments, observational studies and simulation modelling – are useful for achieving a better understanding of the relationships between fire and biota. The value of links between various sorts of studies was neatly summarised by Dooley and Bowers (1998, p. 969) who argued that experiments were: ‘an important intermediary between the inherent abstraction of simulation modelling and what is observed in the real world’.

11.9 Ecological theories, fire disturbance and biodiversity conservation

Several areas of ecological thinking are relevant to fire management and attempts to conserve biodiversity, and three of these are examined here: (1) the biological legacies concept, (2) creating greater congruence between human and natural disturbance regimes, and (3) the intermediate disturbance hypothesis.

The biological legacies concept and biodiversity

Media descriptions of fires often report that ‘x hectares of land have been destroyed’. The reality is that forests, woodlands and other vegetation types are rarely, if ever, completely destroyed by fire. Patches of vegetation actually remain intact within the broad boundaries of a
fire (Delong and Kessler, 2000; Mackey et al., 2002). Within burned areas, fires leave behind considerable biological legacies: organisms, organically-derived structures, and organically-produced patterns that persist from the pre-disturbance ecosystem (Franklin et al., 2000; see Section 11.2). The presence, type and number of biological legacies remaining after fire affects the persistence of populations and the trajectory of succession and recovery (Turner et al., 1998). The concept of ‘destroyed’ forest and woodland is particularly inappropriate in Australia, because many species have evolved strategies for surviving fire (see Box 11.5).

Biological legacies have a range of important ecological functions. They can:

- survive, persist and regenerate after disturbance and be incorporated as part of the recovering stand
- assist other species in persisting in a disturbed area through a variety of mechanisms (often termed a ‘life-boating’ function)
- provide habitat for species that eventually recolonise a disturbed site – a phenomenon referred to as ‘structural enrichment’ of the post-disturbance area (Lindenmayer and Franklin, 2002)
- influence patterns of recolonisation in the disturbed area; for example, biological legacies within a disturbed area can provide foci that facilitate population recovery; that is, recovery can occur not only via colonisation from neighbouring disturbed areas, but also from organisms and structures persisting within a disturbed area (Spies and Turner, 1999)
- provide a source of energy and nutrients for other organisms; this function is particularly important as it relates to maintaining a flow of energy into the soil to maintain soil organisms, including mycorrhizal fungi (Hooper et al., 2000)
- modify or stabilise environmental conditions in the recovering stand (Perry, 1994).

The floristic composition of an area is strongly influenced by pre-disturbance vegetation and the form of individuals, seeds and other propagules (Franklin et al., 2000). The abilities of many animal species to persist and recolonise are influenced by biological legacies (Whelan et al., 2002). This has implications for decisions about the intensity, timing and frequency of prescribed burning.

Congruence between human disturbance and natural disturbance: values and limitations

There is a widely held view that impacts of human disturbances on biodiversity are less when these disturbances resemble natural ecological disturbances (Attiwill, 1994a,b; Hunter, 1994). The underlying premise is that organisms are best adapted to the disturbance regimes under which they have evolved (Bergeron et al., 1999; Hobson and Schieck, 1999). Conversely, organisms may be poorly adapted to novel ecosystem disturbances, including those involving different disturbance agents, different frequencies or intensities of disturbance, or new combinations of disturbances (Paine et al., 1998; Lindenmayer and McCarthy, 2002).

Congruence between human disturbance and natural disturbance has been extensively explored in forest management (Hunter, 1993; Lindenmayer and McCarthy, 2002), where similarities and differences
between logging and natural disturbance (particularly fire) are thought to be important (Korpilahti and Kuuluvainen, 2002). Patch types, sizes, shapes and the internal complexity of patches (i.e. biological legacies) in unmanaged landscapes can guide the size, location, spatial arrangement and rotation period of logged areas or coupes and the structures to be retained in wood production landscapes (Franklin, 1993c; Mladenoff et al., 1993; Franklin et al., 2000). In Sweden, forest management systems are based on silvicultural systems that aim to emulate natural disturbance regimes (Figure 11.9). For example, in wetland forest, ravines, and small islands in lakes there is no forestry activity and fire is excluded. Forest on watercourses and flat moist areas are burned on average every 200 years, and selective harvesting is recommended for these forests. Forest in most moist or wet areas, for example, the majority of boreal forest in Sweden, is burned about every 100 years, the same as the recommended rotation length. Logged forest is burnt and seed trees are retained on site. All dry forest land (e.g. Pine forest on sedimentary soils on flat terrain) is burned every 50 years. These areas support trees that survive recurrent low-intensity fires, so controlled burning of the forest is used. Because the fire regimes result in multi-aged stands, the final felling operation includes the retention of seed trees.

**Limitations of the human–natural disturbance congruence approach**

In Sweden, prescribed burning is linked with the frequency of natural fires in the same system. The same ideas could be useful in planning prescribed burning regimes in Australian landscapes. However, natural fire patterns are complicated because fire frequencies and fire sizes are variable (Chou et al., 1993; Gill and McCarthy, 1998). Historically, landscapes were disturbed by indigenous people (King, 1963; Bowman, 1998), making it difficult to determine 'natural' patterns (Hunter, 1996; Keith et al., 2002a). Disturbance regimes also change following long-term climate change (Hiura, 1995). Bergeron et al. (1998) noted that large changes in the fire frequency in the boreal forests of Canada during the Holocene meant that there was no single characteristic fire regime for this system. Finally, few disturbances in modern landscapes are 'natural'. For example, fires are often extinguished to limit threats to human life and property. Despite these problems, natural disturbance regimes are a useful model for management practices.

**Fire and logging**

From time to time, the suggestion appears that logging results in acceptable environmental impacts when the timing and extent of operations mimics natural disturbance regimes (Attiwill, 1994a,b). This notion has been used together with assumptions that logging reduces the likelihood of wildfire to recommend timber harvesting in national parks (Tuckey, 2000).

In fact, logged native forests are no less fire prone than unlogged native forests. Many major fires have occurred in areas that were previously heavily logged, including the 1983 Ash Wednesday fires in Victoria (Smith and Woodgate, 1985) and others from southeastern New South Wales (e.g. Recher et al., 1985) and north-eastern Victoria (Loy, 1993). Logging in some wet forests in East Gippsland appears to have shifted the vegetation community composition toward one more characteristic of drier forest (Mueck and Peacock, 1992), which may be more fire-prone. Similarly, Whelan (1995) noted that clear-felling operations led to dense regrowth of saplings, which created more available fuel than if the forest was not clear-felled. Logging does not protect forests from fire, and national

![Figure 11.9.](image)
parks do not need to be logged to prevent it. This does not mean, however, that reserves should not be managed; this need is widely recognised and well accepted (Gill et al., 1999; Woinarski, 1999).

There is little evidence to support the assertion that logging has the same effects on forests as fire (Lindenmayer et al., 1990; Ough, 2001). There are major differences between logged forests and those burned by wildfire (Lindenmayer and McCarthy, 2002). For example, large trees with hollows are depleted in Australian logged forests, which potentially affects the more than 300 species of animals that require hollows (Gibbons and Lindenmayer, 2002). In contrast, many Australian plants and animals have evolved adaptations to recover from fire (Whelan, 1995), although, as outlined above, changes in the fire regimes in the past few centuries threaten many of them (Leigh et al., 1984; State of the Environment, 2001a; Bradstock et al., 2002).

**Intermediate disturbance hypothesis**

The intermediate disturbance hypothesis predicts highest species diversity at intermediate rates and intensities of disturbance (Connell, 1978; Shea et al., 2004; Figure 11.10). The concept was developed to explain species richness in coral reefs and rainforests (e.g. Rogers, 1993; Aronson and Precht, 1995; Molino and Sabatier, 2001), and it has also been applied in studies of other systems, such as plankton communities (e.g. Wilson, 1990), temperate forests, and prairies (Collins et al., 1995). The idea appears intuitively logical, because relatively few species can survive frequent high-intensity disturbances and most seem likely to be better adapted either to frequent low-intensity or infrequent high-intensity perturbation. However, support for the intermediate disturbance hypothesis is inconsistent (e.g. Collins, 1992; Schwilk et al., 1997; Bascompte and Rodriguez, 2000; Beckage and Stout, 2000), possibly because it is unclear what actually defines a ‘disturbance’ and what are appropriate spatial and temporal scales for testing (reviewed by Shiel and Burslem, 2003). Intermediate rates and intensities of disturbance will vary between vegetation types: high-intensity fires 20–30 years apart may correspond to long inter-fire intervals in heathland but in forests they would be sufficiently frequent to eliminate fire-sensitive trees.

Even if appropriately defined and scaled, the intermediate disturbance hypothesis has limited utility for practical conservation biology. Species diversity is only one measure of species response and often not the one of greatest interest or relevance. Other factors, such as the conservation of individual threatened species or species likely to respond (positively or negatively) in a particular way to a given fire regime, will be of greater concern. The largest and most important effects of fire are often in terms of the relative abundances of taxa (Recher et al., 1985).

As outlined earlier, responses to fire are varied and highly complex (as shown in the curves derived by Whelan, 1995; Gill, 1999; see Figure 11.5). Many species will not be present and/or most abundant at intermediate rates of disturbance. Some species will be most likely to occur only after a prolonged period without disturbance (e.g. old growth forest specialists; see Chapter 2). In heterogeneous landscapes composed of several vegetation types and where a management objective is to maintain biodiversity at a landscape level (such as in national parks), managing for intermediate fire frequencies and intensities may not be appropriate. In this situation, an intermediate disturbance regime would benefit some taxa but disadvantage others. A better approach would be to follow the reasoning of Gill (1999) and apply burning regimes that vary between vegetation types and within a given vegetation type, so that each type is represented by a range of different seral or successional stages, thereby enabling species associated with those stages to persist.

**11.10 Cumulative effects of fire and other disturbance processes**

Organisms in many landscapes often have to contend with the combined effects of two or more types of disturbance. The impacts of two types of disturbance
may be independently unimportant but together create significant challenges for biodiversity conservation (Taylor, 1979; Caughley and Gunn, 1996; Paine et al., 1998). Grazing–burning interactions are more detrimental to plant mortality than either alone (Leigh and Holgate, 1979). For example, in the Kimberleys of north-western Australia, domestic Cattle grazing results in the transport of grass seeds into rainforest remnants that then grow into fuel for fires, thereby changing fire and rainforest patch dynamics (McKenzie and Belbin, 1991).

Fire, grazing, or logging acting individually may not have significant detrimental impacts in the forests of south-eastern Queensland. However, when combined, the recruitment of new trees to replace harvested stems is impaired – young seedlings are either burned by subsequent fires or eaten by domestic livestock (Smith et al., 1992).

The following circumstances illustrate some of the pressures on animal populations, particularly those living in wood production forests. In dry years, populations must contend with drought and high temperatures, which create problems for some species (How et al., 1984; Rubsamen et al., 1984; Smith, 1984b). These conditions can then be followed by wildfire (which can further reduce populations; Keith et al., 2002b). Fire-damaged forests may then be subject to salvage logging, also with possible impacts on forest biota (Lindenmayer and Franklin, 2002; Lindenmayer et al., 2004b). Although studies of such potentially compounding processes are difficult and therefore rare, it seems likely that recovery from the combination of the processes will be slow.

In tropical forests in South America, logging and fire are inextricably linked because logging opens the canopy, creates additional coarse and fine fuels, dries the understory, and promotes the development of fire-prone grasses (Holdsworth and Uhl, 1997). Fire frequency can increase and preclude forest recovery. In addition, fires are not constrained to logged ignition points; Putz et al. (2000) noted that in Bolivia in 1999, the area of logged forest was 10% of the total area burned by fire.

Where introduced weeds displace native flora, fire regimes can change because of the flammability or biomass of the invasive species (Cochrane, 1963), leading to additional effects on native plants (Wardell-Johnson and Nichols, 1991). This phenomenon is of increasing importance in northern Australia (Whitehead and Dawson, 2000; D. Bowman, personal communication), where many exotic pasture grasses have been introduced (Lonsdale, 1994; Low, 1999; see Chapter 7). In particular, some of the grasses introduced from Africa have substantial biomass late in the dry season, and the additional fuel changes fire regimes. Burning in the late dry season may create novel problems for a number of native mammals (Pardon et al., 2003).

As noted in Section 11.6, fire refugia, whether they are individual rocky areas (Whelan et al., 2002) or entire patches of unburned vegetation (Delong and Kessler, 2000; Mackey et al., 2002), can be critical for species survival and post-fire recovery. Populations depending on post-fire refugia may be particularly susceptible to additional impacts, for example bush rock collection or post-fire salvage harvesting (Mackey et al., 2002).

11.11 Fire and reserve design

Three guiding principles – comprehensiveness, adequacy, and representativeness (CAR; Dickson et al., 1997; see Chapters 4 and 16) – underpin reserve design. Lindenmayer and Franklin (2002) argued for a fourth principle: replication. This refers to the need for a reserve system to contain multiple protected areas of a given vegetation type, forest community or species (see Chapter 16). Replication limits the risk that all reserved examples of a vegetation type, population or community will be affected by a single catastrophic event such as a wildfire (Pickett and Thompson, 1978; Lindenmayer and Possingham, 1994).

The need for replication is influenced by the size of a single reserve, especially whether it is large enough to exceed the maximum size of single disturbance event. Large reserves have a better chance of supporting populations that survive natural disturbances such as wildfires (Pickett and Thompson, 1978; Baker, 1992). Large fires are more likely to leave unburned patches within their boundaries and contain a mosaic of post-fire recovery patterns that are needed for a range of biodiversity (Keith et al., 2002a). If some unaffected area remains after a disturbance, propagules or offspring can recolonise disturbed areas.

A recurring theme after major fires is that government agencies are blamed because there has been insufficient fuel management and that, as a result, there should be no additional reserves until they can be ‘managed properly’. In fact, more fires burn into
national parks than out of them (Barnett, 1994; B. Gilligan, personal communication). National parks are set aside for many reasons other than to manage fuels (see Chapter 16). The ability to reduce fuel over vast parts of the protected area network (even if it were desirable) is clearly not feasible (Whelan, 2002). In response, park managers are now increasingly focusing prescribed burning at the boundaries of parks adjacent to human infrastructure (I. Pulsford, personal communication; S. Troy, personal communication).

11.12 The future
Climate shapes the broad-scale distribution patterns of biota, including vegetation types (Woodward, 1987), and it is likely to alter future fire regimes through changes in temperature, rainfall, evaporation and the frequency of lightning strikes (Cary, 2002). Goldammer and Price (1998) reported that lightning frequency could double over much of continental Australia if the current levels of CO2 in the atmosphere were to double. However, predicting the impacts of future climate change on fire regimes will be difficult, particularly for rare catastrophic fire events. In addition, the predictions of global climate models (GCMs) for rainfall, temperature and other meteorological patterns are very uncertain. Regional climate models (RCMs) are even more uncertain. For example, Mackey et al. (2002) were unable to obtain appropriate input data or suitable regional climate models to simulate future fire regimes in the forests of central Victoria.

There is general consensus about the need for more detailed field-based empirical work on the impacts of fire on the Australian biota. For example, Whelan (1995, p. 307) wrote that:

long-term experimental studies [of fire] are badly needed, but are rare.

In the case of particular groups (such as birds), Woinarski (1999, p. 57 and 59) noted:

there are few data on the long-term impacts of a sustained regime of control burning

and

The search for general pattern in response is further hampered by the very variable and limited research effort, with few long-term studies and little experimentation with a range of fire treatments.

Keith et al. (2002a) and Whelan et al. (2002) highlighted the critical need for monitoring and predicting the impacts of fire regimes on biodiversity. The problem of a lack of monitoring pervades all areas of conservation biology. Nevertheless, these statements, together with the number of species threatened by fire in Australia, indicate that a major research and management challenge exists to identify ecologically appropriate fire regimes for different vegetation types and their associated biota (Gill, 1999; Cary et al., 2003).

11.13 Conclusions
Fire has a complex relationship with the Australian biota. The impacts of fire vary between vegetation types and species. The response of biota is influenced by the intensity of fires; the interval between the spatial pattern of fires (including the size, location and condition of unburned areas within the fire boundary); the number, quantity and types of biological legacies left within a disturbed area (Franklin et al., 2000); and the life history attributes of the biota in the area subject to fire (Whelan, 1995; Woinarski, 1999). Other factors make the situation even more complex, including weather patterns, changing climate conditions, altered past burning regimes, reduction of the total overall area of forest and woodland due to land clearing, and the need to protect human lives and assets.

There are no simple or generic fire management guidelines that can be applied uncritically to all species and all landscapes. Some species will remain relatively immune or benefit considerably from more frequent fire, whereas others will be negatively affected (Whelan, 1995; Gill, 1999). Designing fire management strategies that are sympathetic to the conservation of biota requires careful planning, setting objectives and subsequent monitoring. This is important particularly because inappropriate fire regimes have contributed to the extinction of several species of plants and animals and threaten the long-term persistence of many others.

It is perhaps unsurprising that the impacts of fire on the vast majority of Australian organisms remain poorly known. A major and sustained research effort is required to better understand such a key environmental process, particularly given the number of species threatened by fire. Sustaining the research effort is critical because public concern about fires
and fire impacts is usually short-lived following major fire years in southern Australia. The lack of relevant monitoring data and well-designed field experiments hampers efforts to improve our understanding of fire and its impacts. The most obvious shortcoming is the lack of a national fire mapping system (Gill and Bradstock, 2003). As concluded by Cary et al. (2003):

The crucial task of identifying ecologically sustainable fire management policies and practices that are consistent with community safety will not be appropriately tackled until significantly increased levels of research, funding and public interest are maintained in the long term.

11.14 Practical considerations
Fire management for biodiversity conservation is extremely complex – responses to fire will be species-specific, vegetation-type-specific, fire-event specific and even landscape-specific. The best approach is to ensure that the same fire management regime is not applied uniformly everywhere. Variation in the timing, intensity, frequency and spatial location of fires within and between landscapes should reduce the risks of extensive species loss from inappropriate fire regimes (Gill, 1999; Turner et al., 2003). Nevertheless, targeted fire management prescriptions will often be needed to meet the requirements of particular ecological communities. Fire management for conservation must include monitoring of the values that fires are believed to offer.

11.15 Further reading
There are several excellent reviews of fire and the Australian biota. They include Gill et al. (1981, 1999), Whelan (1995), and Bradstock et al. (2002). All of these except Whelan (1995) are edited books, and the chapters in each one give excellent entry points into fire ecology, control, management, and biotic responses. The edited volume by Cary et al. (2003) provides a useful set of papers related to integrated fire, environmental, and policy dimensions in an Australian context. Some of the key findings of the Kapalga Fire Experiment in the savannas of northern Australia – the world’s largest fire experiment – are discussed in the edited book by Andersen et al. (2003). Bowman (1998, 2003) gives an excellent summary of fire in an indigenous context. Richards et al. (1999) gives an elegant example of decision theory modelling to manage fire in a reserve. The State of the Environment Report (2001a) outlines a range of impacts of fire in the Australian environment. Volume 1, issue 5 of the journal Frontiers in Ecology and Environment (June 2005) contains some informed insights into fire regimes and related issues, such as biodiversity conservation by some of the world’s leading fire scientists. Shea et al. (2004) is a valuable review of the intermediate disturbance hypothesis. A major report by the State Government of Victoria (2003) provides a detailed and wide-ranging account of many facets of fire management, control and other issues (including conservation ones) following the inquiry into the 2002–03 bushfires in Victoria. A wide-ranging discussion of fire and fire management and its impacts on Australian birds has been compiled by Olsen and Weston (2005).